BIOLOGICAL EVALUATION OF THE REVISED IDAHO WATER QUALITY STANDARD FOR TEMPERATURE FOR THE SNAKE RIVER BELOW THE HELLS CANYON DAM TO ITS CONFLUENCE WITH THE SALMON RIVER

PREPARED FOR:
U.S. FISH AND WILDLIFE SERVICE AND
NATIONAL MARINE FISHERIES SERVICE

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BE	Biological Evaluation
BO	Biological Opinion
CFR	Code of Federal Regulations
CWA	Clean Water Act
°C	Degrees Celsius
°F	Degrees Fahrenheit
DPS	Distinct Population Segment
EFH	Essential Fish Habitat
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
HCC	Hells Canyon Complex
IDAPA	Idaho Administrative Procedures Act
IDEQ	Idaho Department of Environmental Quality
IDFG	Idaho Department of Fish and Game
LAA	May affect, and Likely to Adversely Affect
MWMT	Maximum of the weekly maximum temperatures for a period of time
NE	No Effect
NLAA	may affect, but Not Likely to Adversely Affect
NMFS	National Marine Fisheries Service
NOAA	National Oceanographic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NSTP	Natural Seasonal Thermal Profile
ODEQ	Oregon Department of Environmental Quality
ODFW	Oregon Department of Fish and Wildlife
PBF	Physical and Behavioral Features
PCE	Primary Constituent Elements
7dadm	Seven-day average of the daily maxima
SR	Snake River
SSC	Site-specific criterion
SRKW	Southern Resident Killer Whales
TMDL	Total Maximum Daily Load
USEPA	US Environmental Protection Agency
USFWS	US Fish and Wildlife Service
WDFW	Washington Department of Fish and Game
WMT	Weekly Maximum Temperature
WQS	Water Quality Standard

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1. BACKGROUND/HISTORY

1.1. Introduction

This Biological Evaluation (BE) prepared by the Region 10 U.S. Environmental Protection Agency (USEPA), addresses the proposed action in compliance with Section 7(c) of the Endangered Species Act (ESA) of 1973, as amended. Section 7 of the ESA assures that, through consultation (or conferencing for proposed species) with the National Marine Fisheries Service (NMFS) and/or the U.S. Fish and Wildlife Service (USFWS), federal actions do not jeopardize the continued existence of any threatened, endangered or proposed species, or result in the destruction or adverse modification of critical habitat.

The federal action that is the subject of this biological evaluation is the USEPA's proposed approval of the revision to Idaho's site-specific criterion (SSC) for temperature for the Snake River, below the Hells Canyon Dam to its confluence with the Salmon River [IDAPA 58-0102-1102]. The purpose of this BE is to assess the effect of the proposed action on species listed as endangered or threatened under the ESA, and their designated critical habitat. This action has the potential to impact the ESA-listed species that occur in the area including bull trout (*Salvelinus confluentus*), steelhead (*Oncorhynchus mykiss*) Chinook salmon (O. *tshawytscha*), sockeye salmon (O. *nerka*), and killer whale (*Orcinus orca*).

1.2. Organization of Biological Evaluation

This biological evaluation follows the suggested format issued by NMFS. The exception is the addition of a section (Section 8) to address Essential Fish Habitat:

Section 1.0 – Background/History

Section 2.0 – Description of the Action and the Action Area. This section describes Idaho's revised water quality standard and rule changes that USEPA proposes to approve (Proposed Action). The ESA-listed species within the action area for the BE are identified and those that could be affected by the proposed action (species of interest) are listed.

Section 3.0 – Listed Species and Critical Habitat in the Action Area. This section describes the species life-history, critical habitat, ESA listing history, current known range, and status for each of the ESA-listed fish species being considered.

Section 4.0 - Environmental Baseline. This section describes effects of past and ongoing human and natural factors leading to the current status of the ESA-listed fish species in the State of Idaho, focusing on impacts related to habitat, harvesting, hydropower, and hatcheries.

Section 5.0 – Effects of the Action. This section includes an analysis of the direct and indirect effects of the proposed action on the species and/or critical habitat and its interrelated and interdependent activities

Section 6.0 – Cumulative Effects, Other Ongoing Environmental Effects, and Uncertainty. Cumulative Effects describes all "non-Federal" actions reasonably certain to occur in the foreseeable future. Includes state, local, private, and tribal actions (e.g. residential developments, watershed enhancement, etc.). Ongoing Environmental Effects includes ongoing environmental conditions and environmental cycles that could result in an additional stressor along with the Agency's Action. Uncertainty includes quantifiable and unquantifiable unknowns that may or may not be accountable for in the Agency's analysis of effects to the species.

Section 7.0 – Conclusions. This section describes the USEPA's effects determination for each of the species addressed as well as critical habitat. The three possible effects determinations for each species are: 1) No Effect (NE); 2) May Affect, but Not Likely to Adversely Affect (NLAA); and 3) May Affect, and Likely to Adversely Affect (LAA).

Section 8.0 – Essential Fish Habitat Analysis. In this section, Essential Fish Habitat (EFH) is assessed for potential adverse impacts resulting from the proposed action.

Section 9.0 Literature cited

1.3. Consultation History

Early coordination and pre-consultation with NMFS and USFWS were conducted during a series of phone conversation including:

Informal phone calls and in-person discussions in October and November, with conference calls November 20, 2018, February 19, 2019, and March 5, 2019 with USFWS; and November 28, 2018, February 14, 2019, February 25, 2019, March 25, 2019, and March 28, 2019 with NMFS.[1] An interim draft Biological Evaluation was shared with the Services on March 12, 2019. Several email exchanges took place after the interim draft was reviewed by the Services.

On December 7, 2018, the USEPA transmitted a letter to NMFS and USFWS with a list of potentially affected listed species relevant to the Agency's Action. The EPA received responses back on December 19, 2018[2] and December 20, 2018[3] from the USFWS and NMFS, respectively, concurring with the USEPA's list of potentially affected listed species.

Idaho Department of Environmental Quality (IDEQ) has been accepted by the USEPA as an applicant for the purposes of formal consultation pursuant to ESA Section 7 on the Agency's Action.

[1] Note a break in communications from December 22, 2018-January 25, 2019 due to an extended government furlough for staff

[2] Communication from Jason Flory, USFWS

[3] Communication from Ritchie Graves, NOAA

2. DESCRIPTION OF THE ACTION AND ACTION AREA

2.1. Action Overview

This section provides a detailed description of the current water quality standard (WQS) and the revised water quality standard submitted to the USEPA by IDEQ that is the focus of the Agency's Action. Background on the existing water quality standard and the process that USEPA uses to review a rule revision to a SSC is also described.

2.2. Description of Action Area

The Action Area for this SSC revision is within a segment of the Hells Canyon Reach of the Snake River. The reach extends from the base of the Hells Canyon Dam downstream approximately 59 miles to the Salmon River/Snake River confluence (Figure 2.1). This includes river miles 247 to 188 of the Snake River. The downstream end of the reach, at the Salmon River Confluence, is about 12 miles upstream of the Washington State border. This portion of the Snake River forms the border between Idaho and Oregon. Oregon WQS for spawning and other criteria to protect cold water fish apply to the reach, extending to the Washington border. Downstream of the Oregon border, this portion of the Snake forms the Washington and Idaho border until it reaches the Clearwater. Idaho and Washington WQS apply to the further downstream reach of the Snake River.

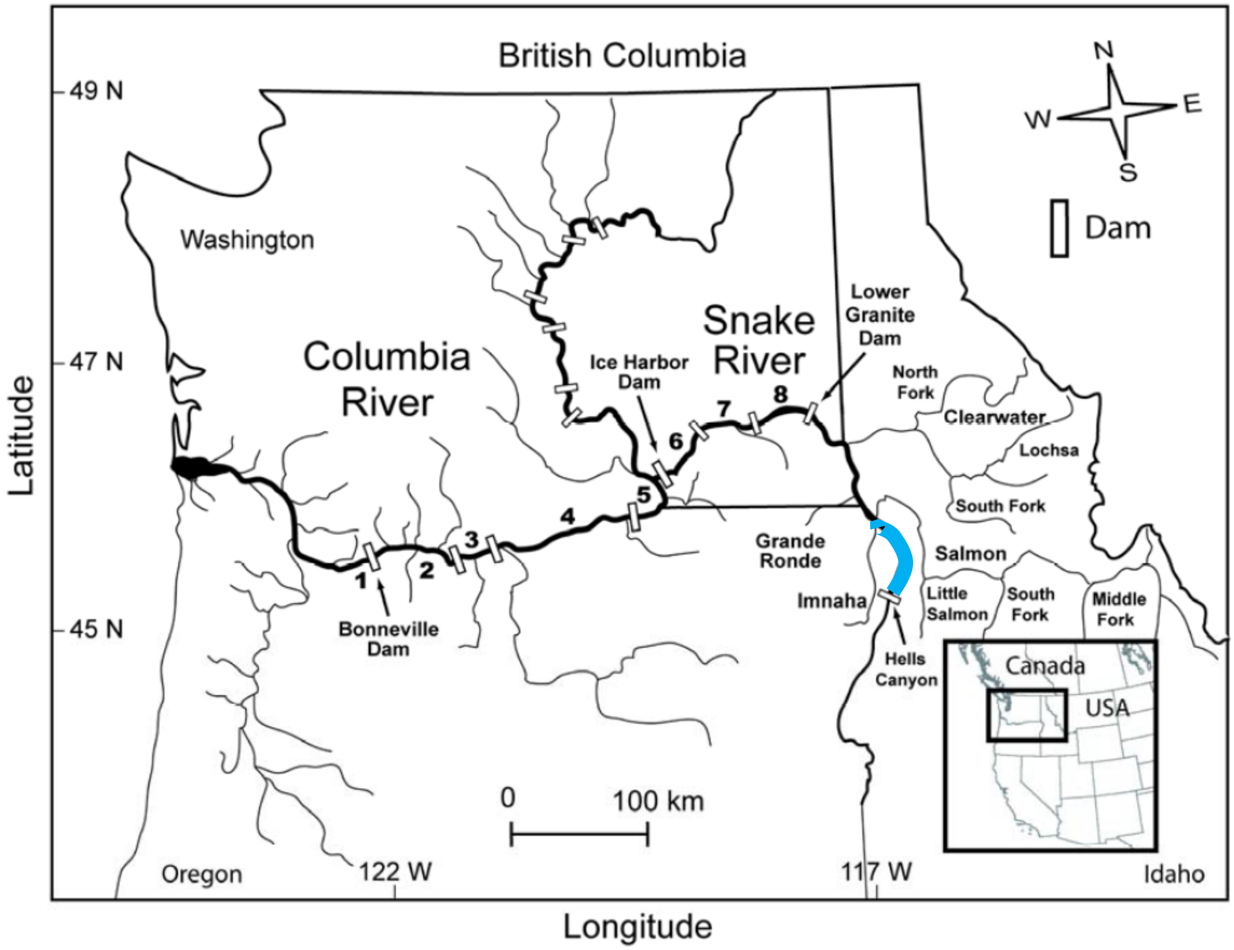

Figure 2.1. Portion of the Hells Canyon Reach of the Snake River included in this action, highlighted in blue. (Adapted from Keefer et al. 2018).

The Snake River is a large river and is the largest tributary to the Columbia River. Its watershed includes areas of Idaho, Nevada, Oregon, Utah, Washington, and Wyoming. The Snake River drains about 87% of the State of Idaho, 17% of the State of Oregon, and greater than 18% of the State of Washington (IDEQ 2004). A significant complex of dams known as the Hells Canyon Complex (HCC) is located on the Snake River, above the Hells Canyon Reach. This complex is comprised of the three dams Brownlee, Oxbow, and Hells Canyon Dams and the associated reservoirs (Figure 2.2). Dworshak Reservoir, on the Clearwater, includes a temperature control structure which is used to pump colder water downstream to the Snake River during peak temperatures to improve migration conditions for salmonids. Downstream of the confluence with the Clearwater River is Lower Granite Dam.

Due to impacts on the Southern Resident Killer Whale's (SRKW) prey base, the action area for SRKW also includes the portion of the eastern Pacific Ocean in which the SRKW feeding areas overlap with Chinook salmon from the Snake River.

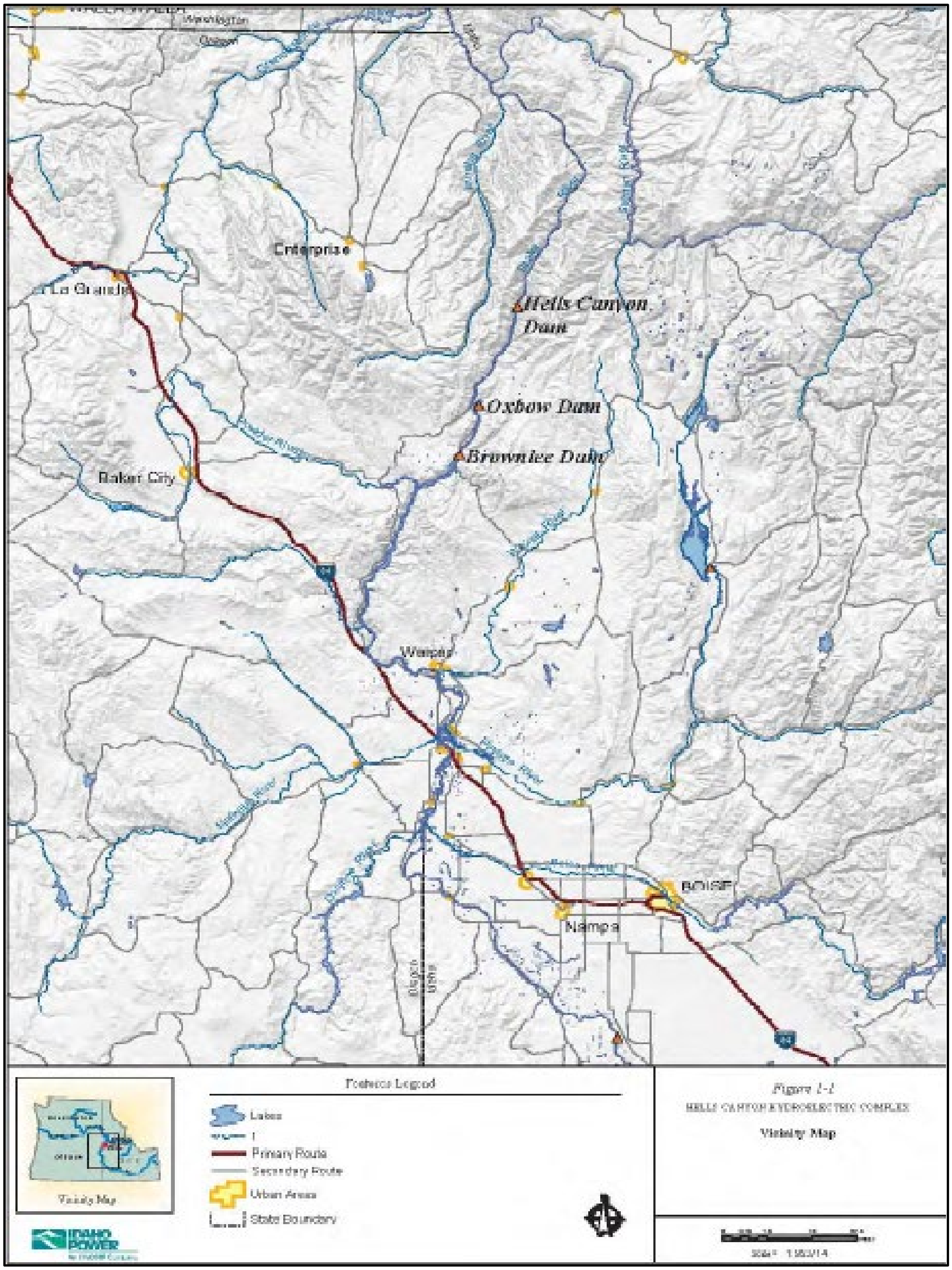

Figure 2.2. Map of the Hells Canyon Complex (Idaho Power Company, 2018)

2.3. Background of Idaho Water Quality Standards and other State Water Quality Standards that Apply to the Action Area

A water quality standard defines the water quality goals for a waterbody by designating the use or uses to be made of the water, by setting criteria necessary to protect the uses, and by preventing or limiting degradation of water quality through antidegradation provisions. The Clean Water Act (CWA) provides the statutory basis for the water quality standards program and defines broad water quality goals. For example, Section 101(a) states, in part, that wherever attainable, waters achieve a level of quality that provides for the protection and propagation of fish, shellfish, and wildlife, and for recreation in and on the water ("fishable/swimmable").

Section 303(c) of the CWA requires that all states adopt water quality standards and that USEPA reviews and takes action to disapprove or approve these standards. Only approved standards are effective under the CWA. In addition to adopting water quality standards, states are required to review and revise standards every three years. This public process, commonly referred to as the triennial review, allows for new technical and scientific data to be incorporated into the standards. The regulatory requirements governing water quality standards are established at 40 Code of Federal Regulations (CFR) Part 131.

The minimum requirements that must be included in the state standards are designated uses, criteria to protect the uses, and an antidegradation policy to protect existing uses, high-quality waters, waters designated as Outstanding National Resource Waters[4]. In addition to these elements, the regulations allow for states to adopt discretionary policies such as allowances for mixing zones and variances from water quality standards. These policies are also subject to USEPA review and action.

Section 303(c)(2)(B) of the CWA requires the states to adopt numeric criteria for all toxic pollutants for which criteria have been published under Section 304(a). USEPA publishes criteria documents as guidance to states. States consider these criteria documents, along with the most recent scientific information, when adopting regulatory criteria.

All water quality standards officially adopted by each state are submitted to USEPA for review and approval or disapproval. USEPA reviews the standards to determine whether the analyses performed are scientifically sound and evaluates whether the designated uses are appropriate and the criteria are protective of those uses. USEPA determines whether the standards meet the requirements of the CWA and USEPA's water quality standards regulations. USEPA then formally notifies the state of these results. If USEPA determines that any such revised or new water quality standard is not consistent with the applicable requirements of the CWA and USEPA's implementing regulations, USEPA is required to specify the disapproved portions and the changes needed to meet the requirements. The state is then given an opportunity to make appropriate changes. If the state does not adopt the required changes, USEPA must promulgate federal regulations to replace those disapproved portions.

4 U.S. Environmental Protection Agency (EPA). 2017. Water Quality Standards Handbook: Chapter 3: Water Quality Criteria. EPA-823-B-17-001. EPA Office of Water, Office of Science and Technology, Washington, DC. Accessed February 2019. https://www.epa.gov/sites/production/files/2014-10/documents/handbook-chapter3.pdf

2.4. Currently effective water quality standards for temperature for the Snake River Hells Canyon Reach and downstream

Idaho temperature criteria for the protection of cold-water aquatic life are a daily maximum temperature not to exceed 22°C, with a maximum daily average temperature of no greater than 19°C. These criteria apply downstream of the HCC. Both Idaho and Oregon currently apply 13°C as a 7-day average of the daily maxima (7dadm) criterion to the Snake River below the Hells Canyon Complex (HCC) (Idaho's is applied as a maximum of the maxima, or Maximum of the Weekly Average Maximum Temperatures, MWMT). The 13°C SSC (which the USEPA previously approved as an SSC for the Snake River from the Hells Canyon Dam to its confluence with the Salmon River) apply from October 23 to April 15. Additional Oregon temperature criteria include a 20°C criterion that applies to migration corridors to protect cold water fish below the HCC, together with narrative criteria that cold water refugia are to be sufficiently distributed to allow for salmon and steelhead migration without significant adverse effects from higher water temperatures elsewhere in the stream, and a seasonal thermal pattern in the Snake and Columbia rivers that reflects the natural seasonal thermal pattern (NSTP). Both Idaho and Oregon WQS include provisions that allow for a *de minimis* increase of 0.3°C to river temperatures when temperatures exceed the criteria (Oregon) or when natural thermal sources result in exceedances upstream (Idaho). Excerpts of relevant applicable standards for Oregon, Idaho, and Washington are below.

2.4.1. Oregon

Table 2.1. Oregon designated uses for the Snake River Basin.

TABLE 121B

BENEFICIAL USE DESIGNATIONS - FISH USES
MAINSTEM SNAKE RIVER

Geographic Extent of Use	**Salmon and Steelhead Migration Corridors (20°C)**	**Redband or Lahontan Cutthroat Trout (20°C)**	**Salmon and Steelhead Spawning through Fry Emergence**
Mainstem Snake River			
Oregon/Washington Border to Hells Canyon Dam (RM 169 to RM 247.5)	X		October 23-April 15
Hells Canyon Dam to Idaho border (RM 247.5 to RM 409)		X	

Table produced November, 2003

OAR 340-041-0028(4)(a)

(a) The seven-day-average maximum temperature of a stream identified as having salmon and steelhead spawning use on subbasin maps and tables set out in OAR 340-041-0101 to 340-041-0340: Tables 101B, and 121B, and Figures 130B, 151B, 160B, 170B, 220B, 230B, 271B, 286B, 300B, 310B, 320B, and 340B, may not exceed 13.0 degrees Celsius (55.4 degrees Fahrenheit) at the times indicated on these maps and tables

OAR 340-041-0028(4)(d)

(d) The seven-day-average maximum temperature of a stream identified as having a migration corridor use on subbasin maps and tables OAR 340-041-0101 to 340-041-0340: Tables 101B, and 121B, and Figures 151A, 170A, 300A, and 340A, may not exceed 20.0 degrees Celsius (68.0 degrees Fahrenheit). In addition, these water bodies must have cold water refugia that are sufficiently distributed so as to allow salmon and steelhead migration without significant adverse effects from higher water temperatures elsewhere in the water body. Finally, the seasonal thermal pattern in Columbia and Snake Rivers must reflect the natural seasonal thermal pattern

2.4.2. Idaho

Cold water and spawning aquatic life criteria:

Idaho applies a cold water life criterion in the summer maximum period (outside of the spawning period) of 22°C daily maximum and 19°C daily average. For spawning, Idaho applies a statewide criterion of 13°C as a daily maximum, and 9°C as a daily average. As described above, the USEPA previously approved an SSC of 13°C as a MWMT for the Snake River below Hells Canyon Dam to its confluence with the Salmon River.

Downstream Protection Provision

A downstream criterion has been submitted, with Agency Action expected in 2019.

IDAPA 580102(070)

(8) Protection of Downstream Water Quality. All waters shall maintain a level of water quality at their pour point into downstream waters that provides for the attainment and maintenance of the water quality standards of those downstream waters, including waters of another state or tribe.

2.4.3. Washington

Washington applies a criterion of 17.5°C as a 7dadm to its waters downstream of the border with Oregon, Idaho, and Washington.

2.4.4. Idaho, Oregon and Washington – 0.3°C Allowance and Antidegradation Policies

All three states, Idaho, Oregon, and Washington may allow a 0.3°C cap on human sources above the numeric criteria and/or natural condition of the waterbody. [5]

In addition, all three states have antidegradation policies that maintain and protect existing uses and the quality of water necessary to protect those uses, and higher quality waters. For existing uses or "Tier 1" uses, antidegradation policies afford some assurance that where a use is identified and is not designated, appropriate application of statewide criteria and other water quality standards to protect existing uses will be implemented. No activity is allowable under the antidegradation policy which would partially or completely eliminate an existing use, whether or not it is designated. CWA actions, including NPDES permits, TMDLs, and 401 certifications must comply with a state's antidegradation policy.

2.5. Description of Specific Idaho Rules USEPA Proposes to Approve

2.5.1. Hells Canyon SSC for Temperature (IDAPA)

The following is an excerpt for Idaho's Administrative Code describing the proposed change to SSC for water temperature in the Hells Canyon action area that is subject to the USEPA's approval.

DOCKET NO. 58-0102-1102

286.SNAKE RIVER, SUBSECTION 130.01, HUC 17060101, UNIT S1, S2, AND S3; SITE-SPECIFIC CRITERIA FOR WATER TEMPERATURE.

~~*A maximum weekly maximum temperature of thirteen degrees C (13C) to protect fall chinook spawning and incubation applies from October 23rd through April 15th in the Snake River from Hell's Canyon Dam to the Salmon River.*~~ Weekly maximum temperatures (WMT) are regulated to protect fall chinook spawning and incubation in the Snake River from Hell's Canyon Dam to the confluence with the Salmon River from October 23 through April 15. Because the WMT is a lagged seven (7) day average, the first WMT is not applicable until the seventh day of this time-period, or October 29. A WMT is calculated for each day after October 29 based upon the daily maximum temperature for that day and the prior six (6) days. From October 29 through November 6, the WMT must not exceed fourteen point five degrees C (14.5°C). From November 7 through April 15, the WMT must not exceed thirteen degrees C (13°C).

The USEPA has identified several aspects of this SSC to protect fall Chinook spawning that would change if the adopted rule is approved by the USEPA. The following are specific changes and explanation of each.

- The addition of "Weekly maximum temperatures (WMT) are regulated" in place of "A maximum weekly maximum temperature…to protect"
  - The weekly maximum temperature is a running mean of daily temperature maxima, which is a change from the original metric, which is a maximum of the

[5] As mentioned previously, EPA expects Idaho to submit a de minimis allowance of 0.3C above the applicable numeric criteria in 2019 for the EPA's review and action. Idaho rules currently include a de minimis provision of an additional 0.3C above the natural temperature of waterbody for point source discharges

weekly maximum temperatures, or the single highest WMT that occurs during a given year or other period of interest (e.g., a spawning period).
    - The additional "are regulated" language is interpreted to mean that the criteria target is met through regulatory means through CWA programs, including TMDLs, 401 certifications, and permitting.
- The change in magnitude of the criterion to 14.5°C as a WMT from 13°C as a MWMT calculated for the period 10/23-11/6. 13°C continues to apply from 11/7 to 4/15 to protect fall Chinook spawning. 13°C is met as a 7dadm starting on 11/7.
- Adds a specification that the criterion is lagged and applies on 10/29 as a start date after averaging seven daily maximum temperatures from 10/23-10/29.

Currently, this segment of the Snake River typically does not meet Idaho and Oregon water temperature criteria at all times and locations. The USEPA believes that implementation of the water quality standards is key to changing the current condition; however, the only action under consideration in this BE is whether the proposed standards themselves and USEPA's approval will have an adverse effect on the species of interest. Finally, the analysis of the effects of the proposed action assumes that ESA-listed species and critical habitat are exposed to waters meeting the proposed water quality standards. This will be explained further in the Effects Analysis (Section 5).

3. LISTED SPECIES & CRITICAL HABITAT IN THE ACTION AREA

This section presents the biological and ecological information for each of the identified to be of concern for this action. The purpose is to inform on the species' life history, its habitat and distribution, and other data on factors necessary for survival to provide background for the analysis. This section also describes the human activities and natural events that have contributed to the current status of the listed species/critical habitat. The information presented is extensively cross-referenced from existing documents including recent status reviews, recovery plans, and biological opinions.

3.1. Occurrence of Listed Species and Critical Habitat in the Action Area

USEPA compiled a list of threatened and endangered species that may occur in the action area (waters at and in the vicinity to which the action applies within the Snake River below the Hells Canyon Dam) from information available on the NMFS and USFWS websites (Table 3.1). USEPA requested and received confirmation of this current ESA species list for this BE from the NMFS and USFWS (received 12/19/18 from Jason Flory USFWS and 12/20/18 from Ritchie Graves NOAA by Rochelle Labiosa USEPA). USEPA staff have conducted conference calls with NMFS and USFWS staff to scope the species and issues that should be the central focus of this ESA consultation. This list (December 2018) of species was then reviewed to determine whether the listed species and/or its designated critical habitat areas would be exposed to the proposed action. A "May Affect" or "No Effect" determination was made for each species.

Table 3.1. Protected Species and Critical Habitat

Protected Species	Scientific Name	Status	Critical Habitat	
			Status	In Action Area
Responsible Agency – NMFS				
Fish				
Snake River fall-run Chinook salmon ESU	*Oncorhynchus tshawytscha*	T	D	X
Snake River spring/summer-run Chinook salmon ESU	*O. tshawytscha*	T	D	X
Snake River Basin steelhead trout (DPS)	*O. mykiss*	T	D	X
Snake River sockeye salmon	*O. nerka*	E	D	X
Responsible Agency - USFWS				
Plants				
Macfarlane's four-o'clock	*Mirabilis macfarlanei*	T		
Spalding's catchfly	*Silene spaldingii*	T		
Fish				
Bull trout	*Salvelinus confluentus*	T	D	X
Mammals				
Canada lynx	*Felis lynx canadensis*	T	D	
Gray wolf	*Canis lupus*	E		
North American Wolverine	*Gulo gulo luscus*	PT		

Key:D=designated, E=endangered, N=not listed,P=proposed, T=threatened

3.2. Species That May Be Affected by This Action

Based on known distribution and life history of ESA species listed above and input from the USFWS and the NMFS, the following threatened and endangered species are considered in this evaluation (Table 3.2). These are all aquatic species that are known or suspected to occur in the state of Idaho in the Snake River below Hells Canyon Dam up to the confluence with the Salmon River, or are dependent upon ESA species that may be affected by this action. These species either reside all or part of their lives in the freshwaters of the Snake River below the Hells Canyon Dam in Idaho or may be affected indirectly by impacts to species that reside therein, and therefore could be directly affected by the surface water quality standards. In addition to the listed salmonid species, SRKW is also considered in this BE. Although the southern resident population of killer whales are a marine species, they are addressed in this BE due to their dependence on Chinook salmon as a prey species. This species is fully analyzed in this BE.

Table 3.2. List of ESA species that may be affected by this action (present in action area or dependent on ESA species in the action area) and addressed in this BE.

Species	ESU/DPS Name	Status (T, E) and Federal Register Notice	Critical Habitat Designated, Federal Register Notice	Managing Agency
Bull trout (*Salvelinus confluentus*)	Columbia River DPS	Threatened 75 FR 63973 (10/18/10)	Designated	USFWS
Steelhead (*Oncorhynchus mykiss*)	Snake River Basin	Threatened 71 FR 834 (01/05/06)	Designated	NMFS
Chinook salmon (*O. tshawytscha*)	Snake River Spring/ Summer Chinook salmon	Threatened 70 FR 37160 (06/28/05)	Designated	NMFS
Chinook salmon (*O. tshawytscha*)	Snake River Fall-Run Chinook Salmon	Threatened 70 FR 37160 (06/28/05)	Designated	NMFS
Sockeye salmon (*O. nerka)*	Snake River Sockeye Salmon	Endangered 70 FR 37160 (06/28/05)	Designated	NMFS
Killer Whale (*Orcinus orca*)	Southern Resident DPS	Endangered 70 FR 69903 (11/18/05)	Designated*	NMFS

**Designated critical habitat does not include the action area.*

3.3. Other ESA Species of the Action Area --"No Effect" Species

The listed plants and mammals in Table 3.1 are not likely to be directly affected by Idaho's criteria revision for temperature. The primary exposure of these species to water quality impacts is through either drinking water or habitat degradation. Neither of these exposure routes is likely to be significantly affected by the proposed changes in the temperature criteria. Based on the pre-consultation conference calls with the USFWS and the NMFS, it was determined that these non-aquatic species would not be directly impacted by changes to the SSC for temperature for the Snake River below the Hells Canyon Dam to its confluence with the Salmon River and thus the approval of the revised SSC would not have an adverse effect on these species. Therefore, USEPA's proposed approval of Idaho's revised temperature criteria will result in a 'No Effect' to MacFarlane's four-o'clock, Spalding's catchfly, Canada lynx, gray wolf, and North American wolverine. These non-aquatic species are not addressed further in this BE.

3.4. Status of the Species and Critical Habitat in the Action Area

The following sections describe each of the ESA-listed salmonid species (fall Chinook salmon, and spring/summer Chinook salmon (*O. tshawytscha*), Snake River steelhead trout (*O. mykiss*), Snake River sockeye salmon (*O. nerka*), bull trout (*Salvelinus confluentus*) and killer whales (*Orcinus orca*) relevant to this action (see Table 3.2). A discussion of the life history, habitat use, and habitat concerns, as well as specific information on occurrence in and use of the action area is presented for each species.

Additionally, the designated critical habitats for these five of species are described. From NOAA's 2015 Opinion: "Interior Columbia Recovery Domain. Critical habitat has been designated in the IC recovery domain, which includes the Snake River Basin, for SR spring/summer-run Chinook salmon, SR fall-run Chinook salmon, UCR spring-run Chinook salmon, SR sockeye salmon, MCR steelhead, UCR steelhead, and SRB steelhead. Major tributaries in the Oregon portion of the IC recovery domain include the Deschutes, John Day, Umatilla, Walla Walla, Grande Ronde, and Imnaha rivers. Habitat quality in tributary streams in the IC recovery domain varies from excellent in wilderness and roadless areas to poor in areas subject to heavy agricultural and urban development (Wissmar et al. 1994; NMFS 2009a). Critical habitat throughout much of the IC recovery domain has been degraded by intense agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer stream flows, impaired water quality, and reduced habitat complexity are common problems for critical habitat in developed areas. Migratory habitat quality in this area has been severely affected by the development and operation of the dams and reservoirs of the Federal Columbia River Power System (FCRPS) in the mainstem Columbia River, Bureau of Reclamation tributary projects, and privately-owned dams in the Snake and Upper Columbia river basins."

In general, the ESA-listed species and their designated critical habitat have been dramatically affected by the development and operation of the Federal Columbia River Power System (FCRPS). In the Columbia River Basin, anadromous salmonids, especially those above Bonneville Dam, have been dramatically affected by the development and operation of the Federal Columbia River Power System (FCRPS) and private hydropower complexes. Storage dams have eliminated spawning and rearing habitat and have altered the natural hydrograph of the Snake and Columbia Rivers, decreasing spring and summer flows and increasing fall and winter flows. Power operations cause flow levels and river elevations to fluctuate, affecting fish movement through reservoirs and riparian ecology, and stranding fish in shallow areas. The eight dams in the migration corridor of the Snake and Columbia Rivers alter smolt and adult migrations. Smolts experience a high level of mortality as they pass through the dams. The dams also have converted the once-swift river into a series of slow-moving reservoirs, slowing the smolts' journey to the ocean and creating habitat for predators. Water velocities throughout the migration corridor now depend far more on volume runoff than prior to the emplacement of mainstem reservoirs.

The analysis of critical habitat is based on the biological requirements of the Action Area related to listed species are those physical or biological features that are essential to conservation of the species. NMFS-USFWS regulations state that federal agencies must consider those physical and biological features that are essential to the conservation of a given species (FR vol.71, no.229, 69060). These features of Critical Habitat are called "primary constituent elements" (PCEs) that are essential to support one or more of the life stages of salmon and steelhead. These PCEs have been changed to 'Physical and Behavioral Features' (PBFs) and will be referred to as PBFs in this BE. The PCEs for the four salmon species assessed in this BE; SR Fall Chinook, SR Spring/Summer Chinook, SR Steelhead, and SR Sockeye, are compiled here in Table 3.3. These species have some level of geographic overlap and have similar life history characteristics and, therefore, require many of the same habitat functions provided by critical habitat. The PBFs for bull trout are presented in the bull trout Critical Habitat description (Section 3.8.5) and the PBFs for SR Killer Whales are in section 3.9.5. The PBFs will be used in the evaluation elements of critical habitat for each species addressed in this BE.

Table 3.3. Salmon and steelhead PBFs of critical habitats and corresponding species life history events.

Primary Constituent Elements for SR spring/summer run Chinook salmon, SR fall-run Chinook salmon, and SR sockeye salmon		
Site	**Site Attribute**	**Species Life History Event**
Spawning and juvenile rearing areas	Access (sockeye) Cover/shelter Food (juvenile rearing) Riparian vegetation Space (Chinook) Spawning gravel Water quality Water temperature (sockeye) Water quantity	Adult spawning Embryo incubation Alevin development Fry emergence Fry/parr growth and development Fry/parr smoltification Smolt growth and development
Juvenile migration corridors	Cover/shelter Food Riparian vegetation Safe passage Space Substrate Water quality Water quantity Water temperature Water velocity	Fry/parr seaward migration Smolt growth and development Smolt seaward migration
Adult migration corridors	Cover/shelter Riparian vegetation Safe passage Space) Substrate Water quality Water quantity Water temperature Water velocity	Adult sexual maturation Adult "reverse smoltification" Adult upstream migration Kelt (steelhead) seaward migration
Primary Constituent Elements for Steelhead		
Freshwater spawning	Spawning gravel /substrate Water quality Water quantity	Adult spawning Embryo incubation Alevin development
Freshwater rearing	Flood plain connectivity Forage Natural cover Water quality Water quantity	Fry emergence Fry/parr growth and development
Freshwater migration	Free of artificial obstructions Natural cover Water quality Water quantity	Adult sexual maturation Adult upstream migration Kelt (steelhead) seaward migration Fry/parr seaward migration

3.5. Species Overview --Chinook Salmon (*O. tshawytscha)*

Chinook salmon, also called king salmon, are the largest and least abundant species of Pacific salmon (NMFS 2005). Chinook salmon are anadromous and semelparous, meaning adults migrate from a marine environment into their natal freshwater streams (anadromous) where they spawn and die (semelparous). Adult female Chinook will prepare a spawning bed, called a redd, in a stream area with suitable gravel composition, water depth, and velocity. Redds will vary widely in size and in location within the stream or river. After laying eggs in a redd, adults will guard the redd from 4 to 25 days before dying. Eggs hatch, depending upon water temperatures, between 90 to 150 days after deposition. Stream flow, gravel quality, and silt load all significantly influence the survival of developing Chinook salmon eggs. Juvenile Chinook may spend from 3 months to 2 years in freshwater after emergence and before migrating to estuarine areas as smolts, and then into the ocean to feed and mature.

Adults spend one to six years in the ocean before migrating back to natal freshwater streams to spawn and subsequently die. Compared to other Pacific salmon species, Chinook prefer larger and deeper stream habitat (NMFS 2005). Juveniles feed on terrestrial and aquatic invertebrates, while subadults (i.e., post-smolt stage) and adults consume larger prey such as shrimp, squid, and small fish (e.g., herring [*Clupea* spp.] and sand lance [*Ammodytidae* spp.]) (Scott and Crossman 1973). The distribution of Chinook salmon in the marine environment is not well characterized; however, they may be found as far north as Alaska, as far south as California, and as far west as Russia and Japan (NMFS 2016). The following summary is taken from USFWS 1998:

Among Chinook salmon two distinct races have evolved. One race, described as a "stream-type" Chinook, is found most commonly in headwater streams. Steam-type Chinook salmon have a longer freshwater residency and perform extensive offshore migrations before returning to their natal streams in the spring or summer months. The second race is called the "ocean-type" Chinook, which is commonly found in coastal streams in North America. Ocean-type Chinook typically migrate to sea within the first 3 months of emergence, but they may spend up to a year in freshwater before emigrating. They also spend their ocean life in coastal waters. Ocean-type Chinook salmon return to their natal streams or rivers in spring, winter, fall, summer, and late-fall runs, but summer and fall runs predominate. both genetic and morphological differences are found between these life history types.

Juvenile stream- and ocean-type Chinook salmon have adapted to different ecological niches. Ocean-type Chinook salmon tend to use estuaries and coastal areas more extensively for juvenile rearing. Stream-type juveniles are much more dependent on freshwater stream ecosystems because of their extended residence in these areas. A stream-type life history may be adapted to those watersheds, or parts of watersheds, which are more consistently productive and less susceptible to dramatic changes in water flow, or which have environmental conditions that would severely limit the success of sub-yearling smolts. At the time of saltwater entry, stream-type (yearling) smolts are much larger than their ocean-type (sub-yearling) counterparts and are, therefore, able to move offshore relatively quickly.

Chinook salmon stocks exhibit considerable variability in size and age of maturation. The relationship between size and length of migration may also reflect the earlier timing of river entry and the cessation of feeding for Chinook salmon stocks that migrate to the upper reaches of river systems. Body size, which is correlated with age, may be an important factor in migration and redd construction success. Under high-density conditions on the spawning ground, natural selection may produce stocks with exceptionally large returning adults.

Early researchers recorded the existence of different temporal "runs" or modes in the migration of Chinook salmon from the ocean to freshwater. Freshwater entry and spawning timing are believed to be related to local temperature and water flow regimes. Seasonal "runs" (i.e., spring, summer, fall, or winter) have been identified based on when adult Chinook salmon enter freshwater to begin their spawning migration. However, distinct runs also differ in the degree of maturation at the time of river entry, the thermal regime and flow characteristics of their spawning site, and their actual time of spawning. Egg deposition must occur at a time that will ensure that fry emerge during the following spring when the river or estuary productivity is sufficient for juvenile survival and growth. The Columbia River supports the freshwater phase of substantial Chinook populations.

NMFS recognizes six ESA-listed Evolutionarily Significant Units (ESUs) of Chinook salmon that spawn in Washington, Oregon, and Idaho. As stated in the previous section, only Chinook ESUs that occupy the Snake River Basin are included in this BE. The Snake River Basin drains an area of approximately 280,000 km^2 and incorporates a range of vegetative life zones, climatic regions, and geological formations. The geographic extent of the Snake River ESU includes the mainstem river and all tributaries, from their confluence with the Columbia River to the Hells Canyon Dam complex.

Chinook salmon runs of the Snake River basin are separated into two ESUs: fall-run and spring/summer run, based on genetic distinction (Waples et al. 1991). Also, the spring/summer run and fall run subpopulations are distinguished from one another by the seasons during which they return to freshwater streams. The characteristics of two ESUs are discussed separately in the following sections. Note, these ESUs may include both naturally spawned and artificially propagated (hatchery stock) fish.

3.5.1. Snake River Fall-run Chinook Salmon

The Snake River fall Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653), and the threatened status was reaffirmed in 2005 (70 FR 37160). In 2016, NMFS conducted a 5-year review of the status of the species and announced a 12-month finding on a petition to delist the species. Based on the best available scientific information, NMFS determined that the "threatened" classification remained appropriate (NMFS 2016; 81 FR 33469).

3.5.1.1. Distribution

The Snake River fall Chinook salmon ESU occupies the Snake River basin, which drains portions of southeastern Washington, northeastern Oregon, and north/ central Idaho (Figure 3.1). The Snake River fall Chinook salmon ESU includes one extant population of fish spawning in the mainstem of the Snake River and the lower reaches of several major tributaries including the Tucannon, Grande Ronde, Clearwater, Salmon, and Imnaha rivers. The ESU also includes four artificial propagation programs: the Lyons Ferry Hatchery and the Fall Chinook Acclimation Ponds Program in Washington; the Nez Perce Tribal Hatchery in Idaho; and the Oxbow Hatchery in Oregon and Idaho (70 FR 37160). Historically, this ESU also included a large population that spawned in the mainstem of the Snake River upstream of the Hells Canyon Dam complex, which is currently an impassable barrier to migration (NOAA Fisheries 2015).

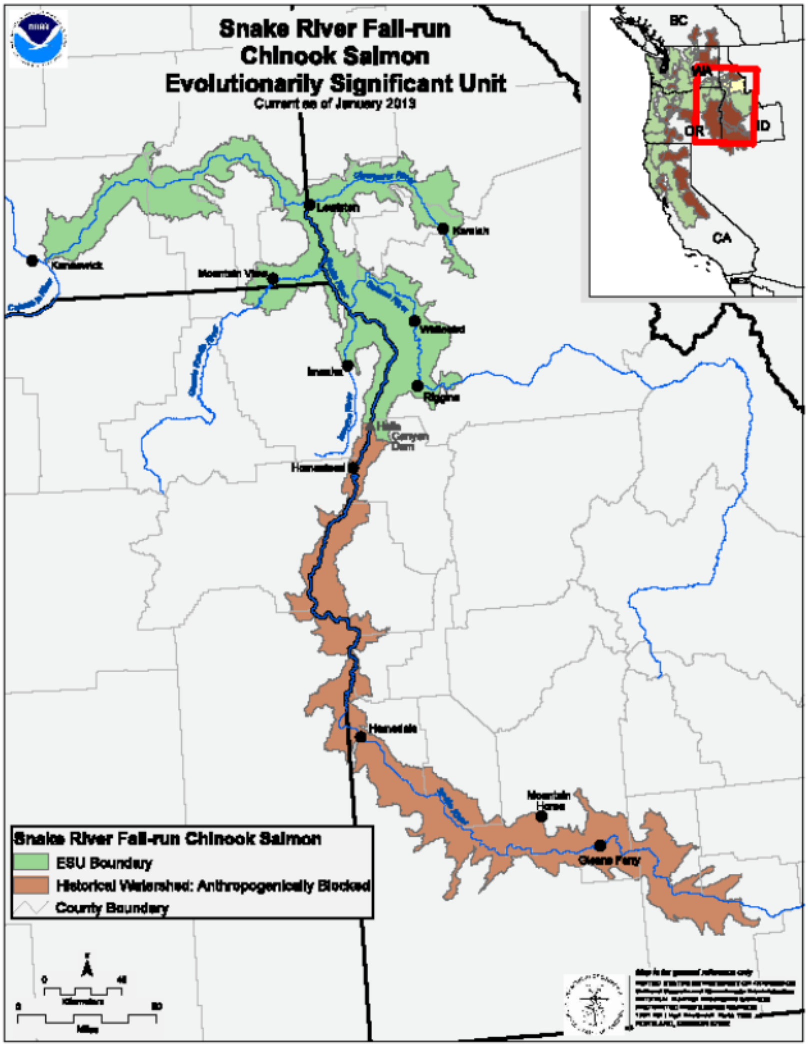

Figure 3.1. Map of fall Chinook ESU (Source: NOAA Fisheries, West Coast Region Species Maps and Data. https://www.westcoast.fisheries.noaa.gov/maps_data/Species_Maps_Data.html).

3.5.1.2. Life history

Snake River fall Chinook salmon enter the Columbia River in July and August and migrate past the lower Snake River mainstem dams from August through November. Spawning takes place from October through early December in the mainstem of the Snake River, primarily between Asotin Creek and Hells Canyon Dam, and in the lower reaches of several of the associated major tributaries including the Tucannon, Grande Ronde, Clearwater, Salmon, and Imnaha rivers (Connor and Burge 2003; Ford et al. (2011)). On their upstream migration adults make extensive use of cold-water patches (refuge) often at tributary confluences (Keefer et al. 2018). Fall Chinook salmon tend to use large, lower elevation streams or mainstem areas. Spawning has occasionally been observed in the tailrace areas of the four mainstem dams (Dauble et al. 1999).

Juveniles emerge from the gravels in March and April of the following year and move seaward slowly as subyearlings.

Until relatively recently, Snake River fall Chinook were assumed to follow an "ocean-type" life history (Dauble and Geist 2000; Good et al. (2005); Healey 1991; NMFS 1992) where they migrate to the Pacific Ocean during their first year of life, normally within three months of emergence from spawning substrate (as young-of-year smolts), to spend their first winter in the ocean. Ocean-type Chinook salmon juveniles tend to display a "rear as they go" strategy in which they continually move downstream through shallow shoreline habitats during the first summer and fall until they reach the ocean by winter (Connor and Burge 2003; Coutant and Whitney 2006). However, a substantial number Snake River fall Chinook juvenile exhibit a "reservoir-type" life history by which they begin their seaward migration later than ocean-types, arrest their migration and overwinter in reservoirs on the Snake and Columbia Rivers, then resume migration, entering the ocean in early spring as age-1 smolts (Connor and Burge 2003; Connor et al. (2002); Connor et al. 2005; Hegg et al. 2013). Analysis of fish scales taken from non-hatchery, adult, fall-run Chinook salmon indicate that approximately half of the returns passing Lower Granite Dam are reservoir type Snake River fall Chinook and overwintered in freshwater (Ford et al. 2011). Tiffan and Connor (2012) showed that young-of-year fish favor water less than 1.8 m deep.

3.5.1.3. Stressors and threats

With hydrosystem development, the most productive areas of the Snake River Basin are now inaccessible or inundated. The upper reaches of the mainstem Snake River were the primary areas used by fall-run Chinook salmon, with only limited spawning activity reported downstream from river kilometer (Rkm) 439. The construction of Brownlee Dam (1958; Rkm 459), Oxbow Dam (1961; Rkm 439), and Hells Canyon Dam (1967; Rkm 397) eliminated the primary production areas of Snake River fall-run Chinook salmon. There are now 12 dams on the mainstem Snake River, and they have substantially reduced the distribution and abundance of fall-run Chinook salmon (Irving and Bjornn 1981). Beyond this major perturbation there are numerous stressors that impact this ESU to Snake River fall Chinook salmon include commercial and recreational harvest, bycatch, and natural predation; reduced habitat and prey quality and quantity; and impeded migration pathways. Within the Snake River Basin, the following stressors impact this ESU.

Human land use practices: throughout the basin, land management has resulted in streams to becoming straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Spawning and rearing habitat quality in tributary streams in the Snake River varies from excellent in wilderness and roadless areas to poor in areas subject to intensive human land uses (NMFS 2015b). Critical habitat throughout much of the Interior Columbia (which includes the Snake River and the Middle Columbia River; MCR) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, and livestock grazing practices.

Water diversions: have substantially reduced flows of many stream reaches designated as critical habitat in the Snake River basin, (NMFS 2015b). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence et al. 1996).

Water quality: Many stream-reaches designated as critical habitat in the Snake River basin are on the Clean Water Act 303(d) list for impaired water quality (e.g., due to elevated water temperature) (IDEQ 2011). Many areas that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures. Water quality in spawning and rearing areas in the Snake River has also been impaired by high levels of sedimentation and by metal contamination potentially from mine waste (IDEQ 2001, IDEQ and USEPA 2003).

Dam Operation: Development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers has severely degraded migration habitat quality for Snake River fall Chinook salmon (NMFS 2008). Hydroelectric development has modified natural flow regimes in the migration corridor causing higher water temperatures in late summer and fall. Other effects include increased rates of piscivorous predation on juvenile salmon due to changes in fish community structure, increased rates of avian predation on juvenile salmon, and delayed migration for both adult and juveniles. Physical features of dams, such as turbines, also kill migrating fish.

Hatchery fish: The continued straying by nonnative hatchery fish into natural production areas is an additional source of risk.

Climate change: One factor affecting the range-wide status of Chinook salmon, and aquatic habitat at large is climate change. For example, salmon abundance is substantially affected by climate variability in freshwater and marine environments, particularly by conditions during early life-history stages of salmon (NMFS 2008b). Sources of variability include inter-annual climatic variations (e.g., El Niño and LaNiña), longer term cycles in ocean conditions (e.g., Pacific Decadal Oscillation, Mantua et al. 1997), and ongoing global climate change. For example, climate variability can affect ocean productivity in the marine environment and water storage (e.g. snow pack) and instream flow in the freshwater environment. Early life-stage growth and survival of salmon can be negatively affected when climate variability results in conditions that hinder ocean productivity (e.g., Scheuerell and Williams 2005) and/or water storage (e.g., ISAB 2007) in marine and freshwater systems, respectively. Severe flooding in freshwater systems can also constrain salmon populations (NMFS 2008c).

3.5.1.4. Population trend and risk

Snake River fall-run Chinook salmon remained stable at high levels of abundance through the first part of the 20th century, but then declined substantially. Although the historical abundance of fall-run Chinook salmon in the Snake River is difficult to estimate, adult returns appear to have declined by three orders of magnitude since the 1940s, and perhaps by another order of magnitude from pristine levels. Irving and Bjornn (1981) estimated that the mean number of fall-run Chinook salmon returning to the Snake River declined from 72,000 during the period 1938 to 1949, to 29,000 during the 1950s. Further declines occurred upon completion of the Hells Canyon Dam Complex, which blocked access to primary production areas in the late 1950s. Estimated returns of naturally produced adults from 1985 through 1993 range from 114 to 742 fish (NMFS 1995).

For the Snake River fall-run Chinook salmon ESU, NOAA Fisheries estimated that the median population growth rate (lambda) over a base period from 1980 through 1998 ranges from 0.94 to 0.86, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared with that of fish of wild origin (McClure et al. 2000). The Snake River component of the fall Chinook run has been increasing during the past few years due to hatchery and supplementation efforts in the Snake and Clearwater River Basins. In 2002, more than 15,200 fall Chinook were counted past the two lower dams on the Snake River, with about 12,400 counted above Lower Granite Dam. These adult returns are about triple the 10-year average at these Snake River projects (FPC 2003).

The National Marine Fisheries Service included the following summary in their 2004 biological opinion for Consultation on Remand for Operation of the Columbia River Power System (NMFS 2004). In their preliminary analysis of recent returns, Fisher and Hinrichsen (2004) reported that the geometric mean abundance of naturally-produced fall Chinook was 3,462 during 2001-2003, compared to 694 in 1996-2000 (a 398% increase). The slope of the population trend increased 8.0% (from 1.16 to 1.24) when the data for 2001-2003 were added to the 1990-2000 series. These results indicate that at least for the short-term, the population has been increasing. Approximately 64% of the aggregate run at Lower Granite Dam was hatchery fish in 2001-2003, compared to 67% during 1990-2000 (Fisher 2004).

According to NOAA's 2015 Pacific Salmon and Steelhead Status Review Update and NOAA's 2016 5-Year Review of Snake River Salmon and Steelhead, "Overall, while new information indicates an improvement in ESU abundance, uncertainty about population productivity and diversity indicate that the biological risk category has not changed enough since the last status review to achieve the desired viability status of highly viable and support delisting (NWFSC 2015; NMFS 2015c)."

More recently, fall Chinook returns have declined overall (approximately 50% of the 10-year average 2007-2017 in 2017), and SR fall-run returns also reflect this downturn. The following Table 3.4 from Peterson et al. (2018) shows the counts for returning fall Chinook at the Bonneville Dam over the past 20 years.

Table 3.4. Adult returns to Bonneville Dam (Source: Peterson et al. 2018, Table ARD-02).

Adult returns by Year of Ocean Entry[1]

Year	OPIH Coho (adults:smolts)	Bonneville spring Chinook (n)	Bonneville fall Chinook (n)	Klamath River fall Chinook (n est.)
1998	0.0128	178,302	192,793	123,856
1999	0.0227	391,367	400,205	187,333
2000	0.0459	268,813	473,786	160,788
2001	0.0258	192,010	610,075	191,948
2002	0.0399	170,152	583,332	78,943
2003	0.0282	74,038	417,057	65,227
2004	0.0193	96,456	299,161	61,374
2005	0.0238	66,624	161,256	132,131
2006	0.0250	125,543	314,995	70,554
2007	0.0255	114,525	283,691	100,644
2008	0.0461	244,385	467,524	90,860
2009	0.0251	167,097	401,576	101,977
2010	0.0234	158,075	350,083	295,322
2011	0.0092	83,299	952,944	165,025
2012	0.0174	188,078	854,503	160,396
2013	0.0675	220,250	954,140	77,821
2014	0.0131	137,176	440,945	24,582
2015	0.0128	83,616	316,833	31,838[2]
2016	0.0156	87,890	186,862	—
2017	0.0171[2]	—	—	—

[1] Counts of spring and fall Chinook salmon are lagged by 2 years. Return ratios for coho salmon are lagged by 1 year.
[2] Estimate based on jack returns.

The following figure depicts interannual variability in total fall-Chinook returns to the Lower Granite Dam, indicating that a steep decline occurred in 2017 and 2018 (Figure 3.2). Further, the Snake River Fall-run natural origin Chinook have also steeply declined (Table 3.5).

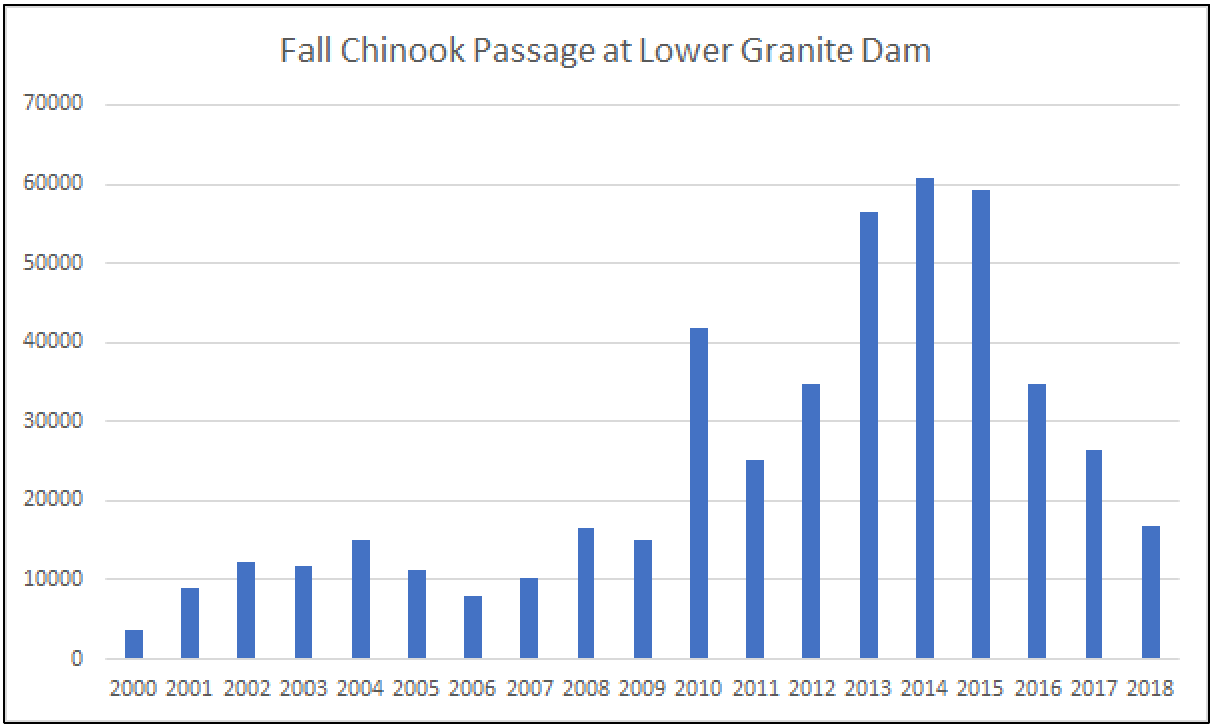

Figure 3.2. Interannual fall Chinook adult return data at Lower Granite Dam. (Source: Columbia River DART)

Table 3.5. Estimated Columbia River return of Snake River natural origin fall Chinook adults 1986-2017(Source: Table 5 in WDFW, ODFW joint status report, 2018).

Year	Columbia River Return	Non-Treaty Zone 1-5 Harvest	Bonneville Dam Count	Treaty Zone 6 Harvest	Non-Treaty Above BON Harvest [1]	Lower Granite Run Size [2]
1986	2,830	652	2,178	723	12	449
1987	1,783	491	1,292	379	2	253
1988	3,558	944	2,614	965	7	368
1989	1,981	373	1,608	608	8	295
1990	508	71	437	169	2	78
1991	1,846	275	1,571	379	17	318
1992	1,289	112	1,178	202	6	549
1993	1,475	107	1,368	270	11	742
1994	958	0	958	173	1	406
1995	1,296	10	1,286	225	9	350
1996	1,729	95	1,634	350	3	639
1997	1,839	99	1,740	459	7	797
1998	730	21	709	165	4	306
1999	2,395	163	2,232	515	11	905
2000	2,612	179	2,432	520	9	1,148
2001	14,133	778	13,355	2,020	63	5,163
2002	3,665	250	3,416	709	11	2,116
2003	8,093	675	7,417	953	33	4,257
2004	8,174	706	7,467	877	21	7,055
2005	9,500	779	8,721	1,434	49	5,299
2006	12,202	928	11,274	2,136	34	4,713
2007	9,878	567	9,311	1,492	64	3,914
2008	8,738	622	8,115	1,615	30	3,937
2009	15,576	1,568	14,008	3,831	53	4,653
2010	12,855	971	11,884	2,141	34	7,302
2011	17,156	2,228	14,928	2,918	53	8,370
2012	19,360	2,641	16,719	3,433	61	12,797
2013	34,669	3,462	31,208	6,429	141	21,124
2014	20,752	2,484	18,268	4,096	32	14,172
2015	24,054	2,530	21,523	4,319	87	16,212
2016	14,493	2,023	12,568	2,907	96	9,772
2017	11,750	1,403	10,997	3,308	86	6,966

[1] *Recent year harvest data for non-treaty recreational fisheries upstream of Bonneville Dam considered preliminary until catch record card data is finalized.*

[2] *Includes release mortalities*

3.5.1.5. Critical habitat

Critical habitat for Snake River fall-run Chinook salmon was designated in 1993 and modified on March 9, 1998 (NOAA 1998) to include the Deschutes River and includes reaches of the Columbia, Snake, and Salmon Rivers and passable tributaries of the Snake and Salmon Rivers (58 FR 68543). The geographic extent of critical habitat is the Snake River to Hells Canyon Dam; Palouse River from its confluence with the Snake River upstream to Palouse Falls; Clearwater River from its confluence with the Snake River upstream to Lolo Creek; North Fork Clearwater River from its confluence with the Clearwater River upstream to Dworshak Dam; and all other river reaches presently or historically accessible within the Lower Clearwater, Hells Canyon, Imnaha, Lower Grande Ronde, Lower Salmon, Lower Snake, Lower Snake–Asotin, Lower North Fork Clearwater, Palouse, and Lower Snake–Tucannon subbasin. SR Fall Chinook PBFs are compiled in Table 3.3.

3.5.1.6. Use of the action area

The bimodal distribution of Chinook salmon counts at Lower Granite Dam show the run timing distinction between the spring-summer and fall Chinook (Figure 3.3), with fall Chinook migrating past LGD in the September/October timeframe. Adult returns above Lower Granite Dam in 2013 exceeded 50,000, of which approximately 21,000 were naturally produced adults.

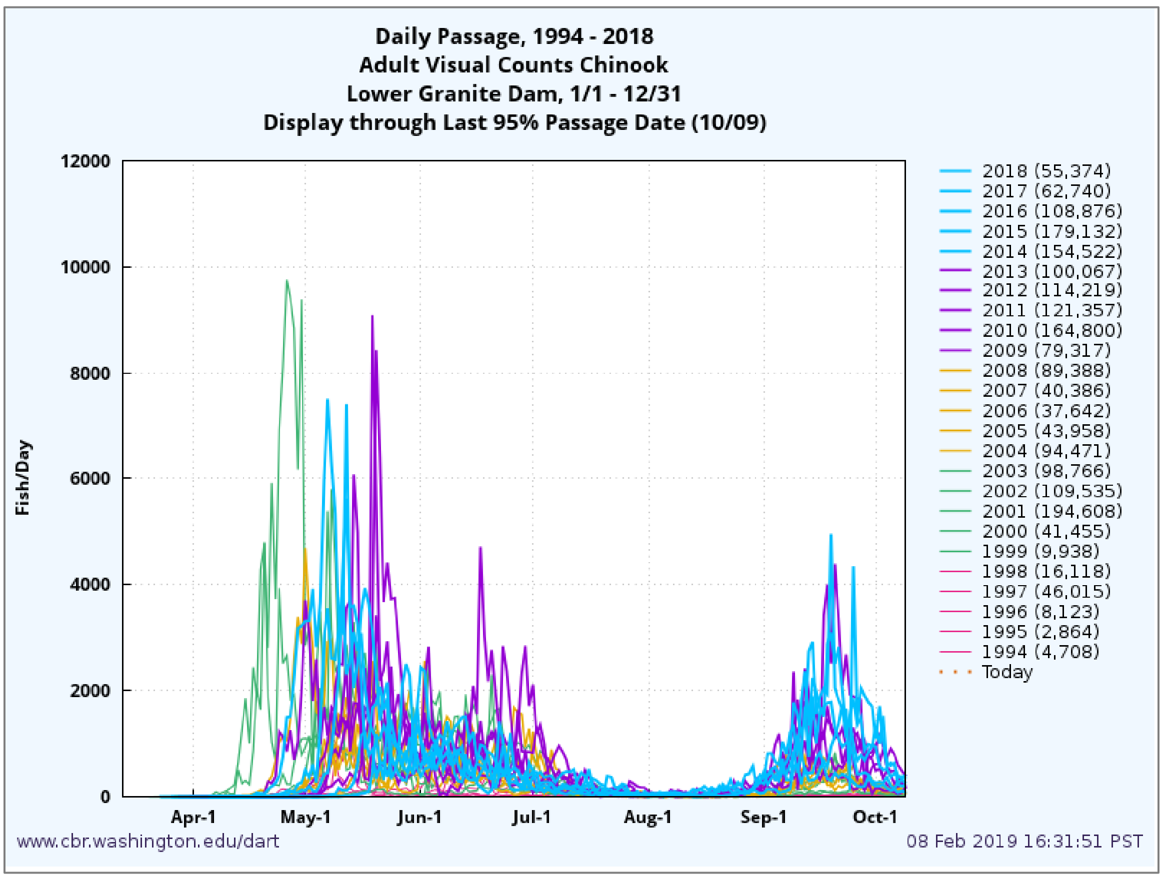

Figure 3.3. Adult Chinook passage counts at Lower Granite dam by year (Source: University of Washington DART data base. Accessed February 8th, 2019). Note bimodal distribution showing the spring-summer and fall runs.

Fall Chinook make extensive use of the Hells Canyon reach for spawning. Redd count data collected from two reaches of the Snake River, below the Salmon River confluence with the

Snake and above the confluence show that the number of redds counted in both reaches has increased steadily beginning in the late 1990s (Figure 3.4). Redds are counted both above and below the Salmon River confluence. A somewhat higher percentage of the total redds are recorded in the reach above the Salmon River confluence, averaging 57% of the total count over the sampling years 1991-2017.

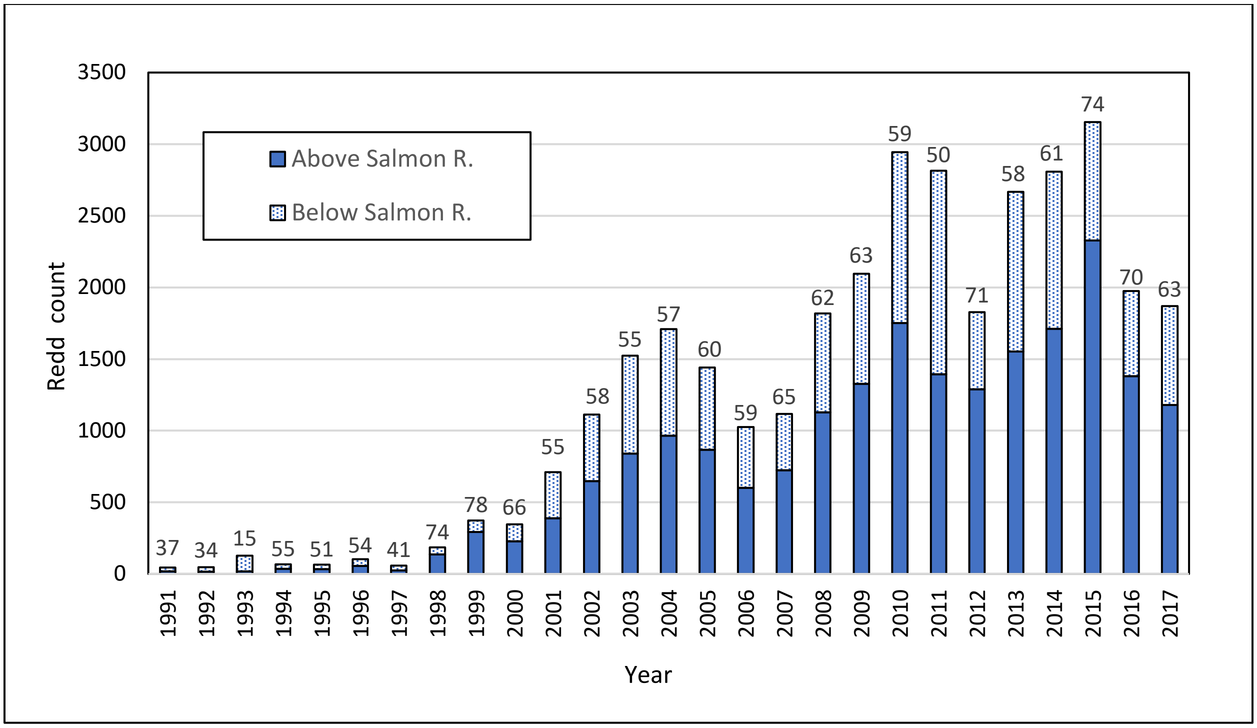

Figure 3.4. Snake River fall Chinook cumulative redd counts from the river reaches above and below the Salmon River confluence. Bar labels indicate percent of total redds from the reach above the confluence (Source: Idaho Power unpublished data. Provided by Brett Dumas to USEPA 01/04/19).

3.5.2. Snake River Spring and Summer Run Chinook Salmon

The Snake River spring/summer Chinook salmon ESU was listed as threatened on April 22, 1992 (57 FR 14653) and the threatened status was reaffirmed in 2005 (70 FR 37160). In 2016, NMFS conducted a 5-year review of the status of the species and based on the best available scientific information determined that the "threatened" classification remained appropriate (NMFS 2016; 81 FR 33468). This ESU includes all naturally spawning populations of spring/summer Chinook in the mainstem Snake River (below Hells Canyon Dam) and in the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (57 FR 23458), as well as the progeny of 15 artificial propagation programs (70 FR 37160). The historical Snake River spring/summer Chinook salmon ESU likely also included populations in the Clearwater River drainage and extended above the Hells Canyon Dam complex; however, current runs returning to the Clearwater River drainages are not considered to be a part of the Snake River spring/summer run Chinook salmon ESU.

3.5.2.1. Distribution

Snake River spring/summer Chinook salmon occupy the Snake River basin in southeastern Washington, northeastern Oregon, and north/central Idaho. Snake River spring/summer-run Chinook salmon are found in several subbasins of the Snake River (CBFWA 1990). Of these, the Grande Ronde and Salmon Rivers are large, complex systems composed of several smaller tributaries that are further composed of many small streams. In contrast, the Tucannon and Imnaha Rivers are small systems with most salmon production in the main river. In addition to these major subbasins, three small streams, Asotin, Granite, and Sheep Creeks, which enter the Snake River between Lower Granite and Hells Canyon Dams, provide small spawning and rearing areas (CBFWA 1990).

3.5.2.2. Life history

Snake River spring/summer Chinook salmon are characterized by their return times. Spring runs are counted at Bonneville Dam beginning in early March and ending the first week of June. Summer runs include Chinook adults that pass Bonneville Dam from June through August. Returning adults will hold migration in deep mainstem and tributary pools until late summer, when they move up into tributary areas to spawn.

In both the Columbia and Snake Rivers, spring-run Chinook salmon tend to spawn in higher-elevation reaches of major Snake River tributaries in mid- to late August, and summer-run Chinook salmon tend to spawn lower in Snake River tributaries in late August and September. The habitats used for spawning and early juvenile rearing also differ among the two runs (Chapman et al. 1991). Summer Chinook are more variable in their spawning habitats; in the Snake River, they inhabit small, high elevation tributaries typical of spring Chinook salmon habitat, whereas in the upper Columbia River they spawn in the larger lower elevation streams characteristic of fall Chinook salmon habitat. The spawning areas of the two runs may overlap. Eggs are deposited in late summer and early fall, incubate through the winter, and hatch between late winter and early spring.

Spring/summer Chinook follow a "stream-type" life history characterized by protracted period of freshwater rearing. Juveniles rear through the summer, and most overwinter and migrate to the sea in the spring of their second year (Healey 1991). Depending on the tributary and the specific habitat conditions, juveniles may migrate extensively from natal reaches into alternative summer-rearing or overwintering areas. Outmigration behavior may differ between the two runs. In both the Upper Columbia and the Snake rivers, spring Chinook salmon migrate swiftly to sea as yearling smolts. Summer Chinook salmon in the Snake River resemble spring-run fish in migrating as yearlings. In the upper Columbia River, they tend to migrate as subyearlings.

Snake River spring/summer Chinook salmon return from the ocean to spawn primarily as four- and five-year-old fish, after two to three years in the ocean. A small fraction of the fish return as three-year old jacks (precocious spawners), of which the majority are males (Good et al. 2005).

3.5.2.3. Stressors and threats

The ability of SR spring/summer-run Chinook salmon populations to sustain themselves through normal periods of relatively low ocean survival remains uncertain. Environmental factors that limit Snake River spring/summer run Chinook salmon are the same as those discussed above for the Snake River fall-run Chinook salmon ESU. Effects related to the hydropower system in the mainstem Columbia River, including reduced upstream and downstream fish passage, altered ecosystem structure and function, altered flows, and degraded water quality are of primary concern. Muir and Williams (2012) noted structural and operational improvements to mainstem Snake and Columbia River hydropower dams in recent years have substantially improved Chinook salmon smolt survival, reduced travel time, and increased connectivity between rearing areas and the Pacific Ocean by restoring entry timing closer to that prior to hydropower development. Despite substantial gains in direct downstream smolt survival and improved upstream passage success through the hydropower system, SAR (smolt-to-adult) return rates have not shown the same improvement in most years. However, variable ocean conditions and increased hatchery production confound comparisons with historical SARs.

Other factors are degradation related to land use (Degradation of floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment) as well as alterations to stream flow, and water quality degradation. Finally, factors that may contribute to depressed and variable SARs (smolt to adult returns) include changes in ocean productivity, increased hatchery production, and the reduction in volume and turbidity of the Columbia River plume due to increased water storage in the basin (from 2015 opinion – Muir and Williams 2012).

3.5.2.4. Population trend and risk

Recent trends in redd counts in major tributaries of the Snake River indicate that many subpopulations could be at critically low levels. Subpopulations in the Grande Ronde River, Middle Fork Salmon River and Upper Salmon River Basins are at particularly high risk. Both demographic and genetic risks would be of concern for such subpopulations, and in some cases, habitat may be so sparsely populated that adults have difficulty finding mates. NOAA Fisheries estimates that the median population growth rate over a base period from 1980 through 1998 ranges from 0.96 to 0.80, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared with the effectiveness of fish of wild origin (Tables B-2a and B-2b in McClure et al. 2000).

The spring Chinook count in the Snake River was at the all-time low of about 1,500 fish as recently as 1995. In 2002, the fish count at Lower Granite Dam was 75,025, which was more than double the 10-year average. Count of both hatchery and wild/natural returns to the Snake River increased in both 2001 and 2002 (FPC 2003). The following summary is from the NMFS 2004 biological opinion for Consultation on Remand for Operation of the Columbia River Power System (NMFS 2004).

'In general, for most of the 24 populations where recent data were available, indices of abundance (i.e., redd counts) for natural-origin SR spring/summer Chinook were high in 2002 and 2003 compared to the 1990s. Fisher and Hinrichsen (2004) provided a preliminary evaluation of the effects of recent natural-origin spring Chinook returns on past geometric mean abundance levels and population trends. The latter were calculated as the slope of the regression line for the (log transformed) index of abundance over time. They assessed whether the geometric mean was greater when calculated from the most recent data (beginning in 2001) compared to a base period (1996-2000) and whether the trend was greater when counts for 2001-2003 were added to the 1990-2000 data series. Their methods were taken from those used by NOAA Fisheries' BRT (2003). The geometric mean for 2001-2003 (33,581) exhibited a 548% increase over the 1996-2000 base period (5,186 fish). The slope of the trend for the natural-origin population increased 17% (from 0.97 to 1.14) when the data for 2001-2003 were added to the 1990-2000 series, reversing the decline and indicating that, at least for the short-term, the natural-origin population has been increasing. Hatchery fish constituted 69% of the return during the recent period compared to an average of 60% during 1990-2000 (Fisher 2004). Even so, natural-origin fish exhibited the substantial increase in numbers described above. Neither the BRT nor the Interior TRT has reviewed Fisher and Hinrichsen (2004) or Fisher (2004).'

Population level status ratings remain at "high" risk of extinction for all major population groups within the ESU. Although recent natural spawning abundance estimates have increased, all populations remain below minimum natural origin abundance thresholds (Ford 2011). Spawning escapements in the most recent years in each series are generally well below the peak returns but above the extreme low levels in the mid-1990s. Relatively low natural production rates and spawning levels below minimum abundance thresholds remain a major concern across the ESU. In NOAA's 2015 Pacific Salmon and Steelhead Status Review Update, it is stated that Spring and Summer Chinook Salmon are likely to become endangered in the future and that the risk level for this population is stable at the threatened level.

3.5.2.5. Critical habitat

Critical habitat for Snake River spring/summer-run Chinook salmon was designated in 1993 and 1999 and includes reaches of the Columbia, Snake, and Salmon Rivers and accessible tributaries of the Snake and Salmon Rivers (58 FR 68543 and 64 FR 57399). The geographic extent of critical habitat includes all Snake River reaches upstream to Hells Canyon Dam; all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Salmon River basin; and all river reaches presently or historically accessible to Snake River spring/summer Chinook salmon within the Hells Canyon, Imnaha, Lower Grande Ronde, Upper Grande Ronde, Lower Snake-Asotin, Lower Snake-Tucannon, and Wallowa subbasins. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 22,390 square miles in Idaho, Oregon and Washington. SR spring/summer Chinook PBFs are compiled in Table 3.3.

3.5.2.6. Use of the action area

Adult spring and summer Chinook salmon destined for the Snake River enter the Columbia River on their upstream spawning migration from February through March. As shown by the bimodal distribution on Figure 3.3, these fish arrive at lower Granite Dam between April through

August. They reach their natal tributaries between June and August and spawning occurs in August and September one to four months after they begin their migration (Status review 1998-Myers).

Fall Chinook begin constructing redds and depositing eggs in the action area beginning in October. Based on weekly redd surveys (2000-2014) conducted above (Figure 3.5) and below (Figure 3.6) the Salmon River confluence, the median annual earliest redds are built on 10/20 (Julian Day 294) above the Salmon River Confluence and 10/21 (Julian Day 295) below the Salmon River Confluence over this 15 year period. Spawning activity continues into early December in most years.

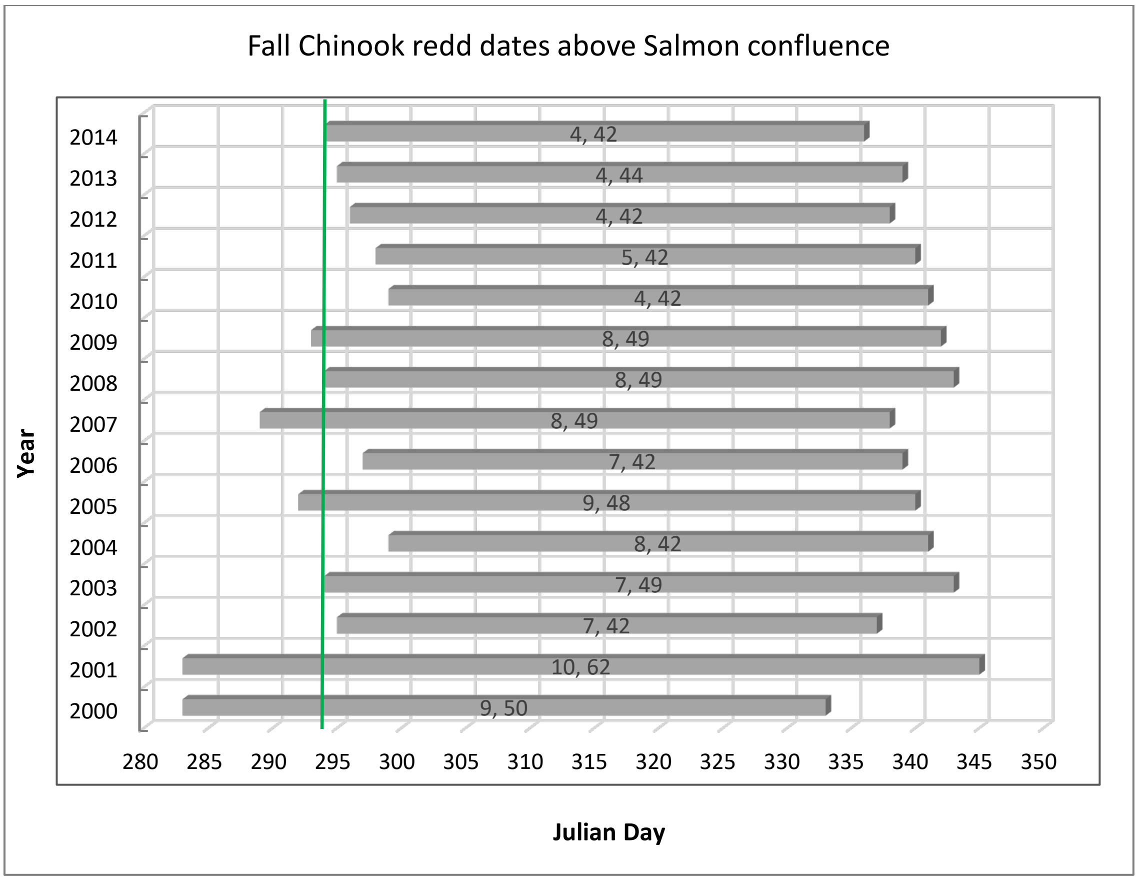

Figure 3.5. Snake River fall Chinook duration of redd observations above the Salmon River confluence in Julian days from earliest and latest non-zero redd observation date, years 2000-2014 (Data Source: Idaho Power unpublished data). Green line shows median (294) earliest Julian date. Labels denote number of non-zero observation events, and duration in days of non-zero redd counts in each year.

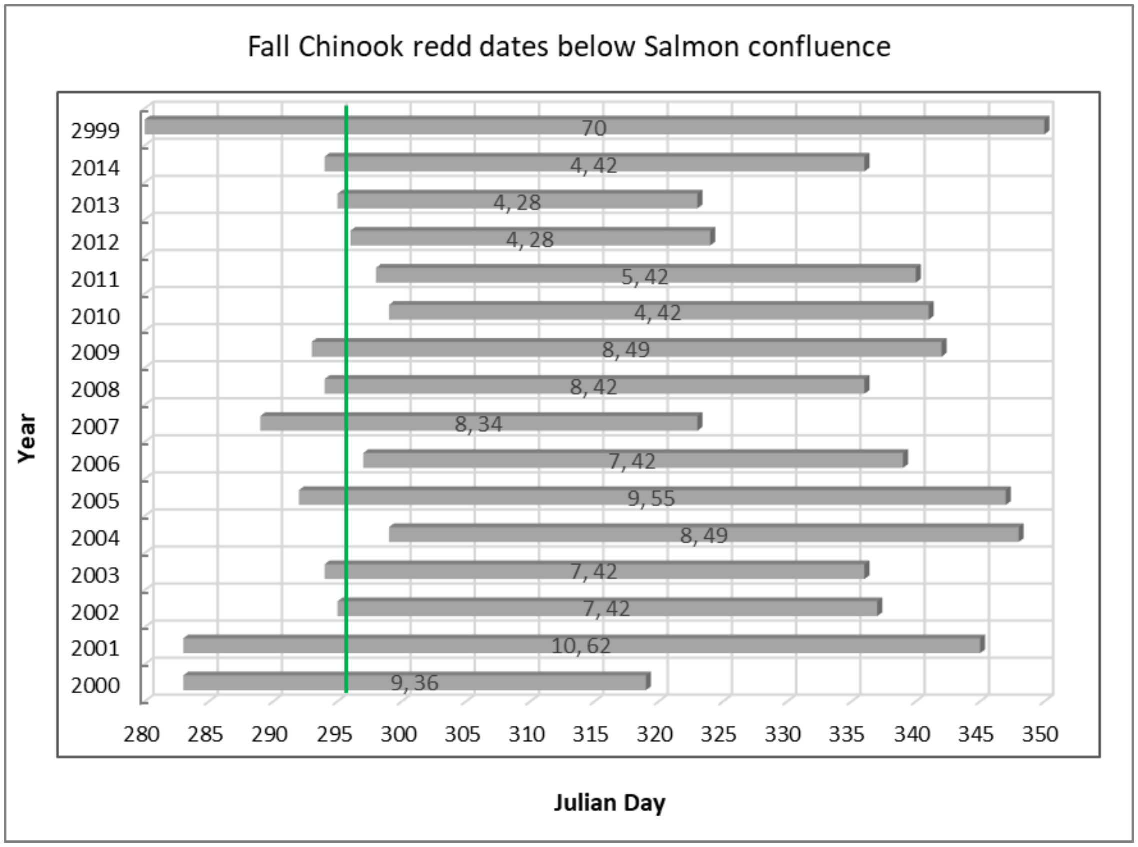

Figure 3.6. Snake River fall Chinook duration of redd observations below the Salmon River confluence in Julian days from earliest and latest non-zero redd observation date, years 2000-2014 (Data Source: Idaho Power unpublished data). Green line shows median (295) earliest Julian date and labels denote number of non-zero observation events, and duration in days in each year.

3.6. Snake River Sockeye Salmon (*O. nerka)*

The Snake River sockeye salmon ESU was first listed as endangered under the ESA in 1991, and the listing was reaffirmed in 2005 (70 FR 37160). On May 26, 2016, in the most recent five-year review for Pacific salmon and steelhead (NMFS 2016), the NMFS concluded that the species should remain listed as endangered (81 FR 33468).

3.6.1. Distribution

This ESU includes all anadromous and resident sockeye salmon from the Snake River basin in Idaho, as well as artificially propagated sockeye salmon from the Redfish Lake captive propagation program. Extant populations of sockeye salmon only occur in the Stanley basin of Idaho. The nonanadromous form (kokanee), found in Redfish Lake and elsewhere in the Snake River Basin, is included in the ESU.

Numbers of Snake River sockeye salmon have declined dramatically over the years. In Idaho, only the lakes of the upper Salmon River (Stanley Basin) remain as potential sources of production. Currently, Snake River sockeye salmon spawn in Redfish, Alturas, and Pettit Lakes (NOAA 2015). The Stanley Basin lakes are located within the Sawtooth National Recreation Area. Basin lakes are glacial-carved and receive runoff from the east side of the Sawtooth and Smoky mountains. All Basin lakes drain to the upper Salmon River which flows into the Snake River and ultimately the Columbia River. Redfish Lake is located approximately 900 river miles from the Pacific Ocean.

3.6.2. Life History

Sockeye salmon are the second most abundant of the seven Pacific salmon species (Quinn 2005). They display more life history diversity than all other members of the Oncorhynchus genus (Burgner 1991). Sockeye salmon are generally anadromous, but distinct populations of non-anadromous O. nerka also exist; these fish are commonly referred to as kokanee (O. nerka kennerlyi). The majority of sockeye populations spawn in or near lakes. Spawning can take place in lake tributaries, lake outlets, rivers between lakes, and on lake shorelines or beaches where suitable upwelling or intra-gravel flow is present. Sockeye fry that are spawned in lake tributaries typically exhibit a behavior of rapid downstream migration to the nursery lake after emergence, whereas lake/beach spawned sockeye rapidly migrate to open limnetic waters after emergence. Lake-rearing juveniles typically spend one to three years in their nursery lake before emigrating to the marine environment (Gustafson et al. 1997). Other life history variants include ocean-type and river-type sockeye. Ocean-type populations typically use large rivers and side channels or spring-fed tributary systems for spawning and emigrate to sea soon after emergence. River-type sockeye rear in rivers for one year before emigrating to sea. Quinn (2005) describes the differences between ocean-type and river-type sockeye as a continuum of rearing patterns rather than as two discrete types.

Upon smoltification, sockeye emigrate to the ocean. Peak emigration occurs in mid-April to early May in southern sockeye populations (generally south of 52ºN latitude) and as late as early July in northern populations (62ºN latitude and north) (Burgner 1991). Typically, river-type sockeye populations make little use of estuaries during their emigration to the marine environment (Quinn 2005). Estuarine habitats may be more extensively used by ocean-type sockeye (Quinn 2005). Upon entering marine waters, sockeye may reside in the nearshore or coastal environment for several months but are typically distributed offshore by fall (Burgner 1991).

Snake River sockeye salmon adults enter the Columbia River primarily during June and July and arrive in the Sawtooth Valley, peaking in August. The Sawtooth Valley supports the only remaining run of Snake River sockeye salmon. The adults spawn in lakeshore gravels, primarily in October (Bjornn et al. 1968). Eggs hatch in the spring between 80 and 140 days after spawning. Fry remain in gravels for three to five weeks, emerge from April through May and move immediately into lakes. Once there, juveniles feed on plankton for one to three years before they migrate to the ocean, leaving their natal lake in the spring from late April through May (Bjornn et al. 1968). Out-migrating juveniles pass Lower Granite Dam (the first dam on the Snake River downstream from the Salmon River) from late April to July, with peak passage from May to late June. Once in the ocean, the smolts remain inshore or within the Columbia River influence during the early summer months. Later, they migrate through the northeast Pacific Ocean (Hart 1973, Burgner 1991). SR sockeye salmon usually spend 2 to 3 years in the Pacific Ocean and return in their fourth or fifth year of life.

3.6.3. Stressors and Threats

After eight hydropower dams on the Columbia and Snake Rivers were finished in the 1970s, Snake River sockeye spawning runs declined dramatically. Natural reproduction of sockeye salmon has been impacted by pollution, habitat loss and degradation, overfishing, and loss of spawning and rearing areas (Good et al. 2005).

Spawning and rearing habitat quality in tributaries of the Snake River varies from excellent in wilderness areas to poor in areas of intensive human land uses (NMFS 2016). Critical habitat throughout much of the Interior Columbia (which includes the Snake River) has been degraded by intensive agriculture, alteration of stream morphology (i.e., channel modifications and diking), riparian vegetation disturbance, wetland draining and conversion, livestock grazing, dredging, road construction and maintenance, logging, mining, and urbanization. Reduced summer streamflows, impaired water quality, and reduction of habitat complexity are common problems for critical habitat in non-wilderness areas. Human land use practices throughout the basin have caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations. In many stream reaches designated as critical habitat in the Snake River basin, streamflows are substantially reduced by water diversions (NMFS 2015b). Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence et al. 1996). Many stream reaches designated as critical habitat in the Snake River basin are on the Clean Water Act 303(d) list for impaired water quality (IDEQ 2014).

Migration habitat quality for Snake River salmon has also been severely degraded, primarily by the development and operation of dams and reservoirs on the mainstem Columbia and Snake Rivers (NMFS 2008). Hydroelectric development has modified natural flow regimes in the migration corridor causing higher water temperatures and changes in fish community structure that have led to increased rates of piscivorous and avian predation on juvenile salmon, and delayed migration for both adult and juveniles. Keefer et al. (2008) also examined current run timing of SR sockeye salmon versus records from the early 1960s and concluded that an apparent shift to earlier run timing recently may reflect increased mortalities for later migrating adults. Physical features of dams such as turbines also kill migrating fish.

3.6.4. Population Trend and Risk

SR sockeye salmon returns to Redfish Lake since at least 1985, when the Idaho Department of Fish and Game (IDFG) began operating a temporary weir below the lake, have been extremely small (1 to 29 adults counted per year). NOAA Fisheries proposed an interim recovery level of 2,000 adult SR sockeye salmon in Redfish Lake and two other lakes in the Snake River Basin (NMFS 1995). Because only 16 wild and 264 hatchery-produced adult sockeye returned to the Stanley River Basin between 1990 and 2000, NOAA Fisheries considers the risk of extinction of this ESU to be very high. In 2002, 52 adult sockeye salmon were counted at Lower Granite Dam

(FPC 2003). NOAA states in their 2015 Opinion on EPA's Action on OR WQS: Although the captive brood program has been successful in providing substantial numbers of hatchery produced O. nerka for use in supplementation efforts, substantial increases in survival rates across all life history stages must occur to re-establish sustainable natural production (Hebdon et al. 2004; Keefer et al. 2008). Overall, although the risk status of Snake River sockeye salmon appeared to improve between 2005 and 2011, we determined, in our 2011 5-year review, that this ESU should retain its "endangered" classification.

3.6.5. Critical Habitat

Critical habitat for Snake River sockeye salmon was designated in 1993 and includes the Columbia, Snake and Salmon Rivers, Alturas Lake Creek, Valley Creek, Stanley Lake, Redfish Lake, Yellowbelly Lake, Pettit Lake, Alturas Lake, and all inlet/outlet creeks to these lakes (58 FR 68543). Watersheds containing spawning and rearing habitat for this ESU comprise approximately 510 square miles in Idaho. The watersheds lie partially or wholly within the counties of Blaine and Custer. SR sockeye salmon PBFs are compiled in Table 3.3.

3.6.6. Presence in the Action Area

Although long-term sockeye population has declined, abundance of Snake River sockeye salmon counted at Lower Granite dam has been higher this past decade (Figure 3.7). Once above the lower Granite Dam, these fish use the Snake River as a migration corridor to reach spawning areas in the Salmon River basin (listed above). The confluence of the Salmon River is the limit of upriver distribution of sockeye as they are not known to use the Snake River above the Salmon River confluence (Hells Canyon action area for this BE) for migration nor for other life history phases. Snake River sockeye adults migrate pass Granite Dam in the Summer/fall with most passing upstream prior to September. Further details from the DART pit-tag database shows that only three PIT tagged adult sockeye (out of 964 or 0.3%) have been detected passing Lower Granite Dam after mid-October in the past 10 years (Figure 3.8).

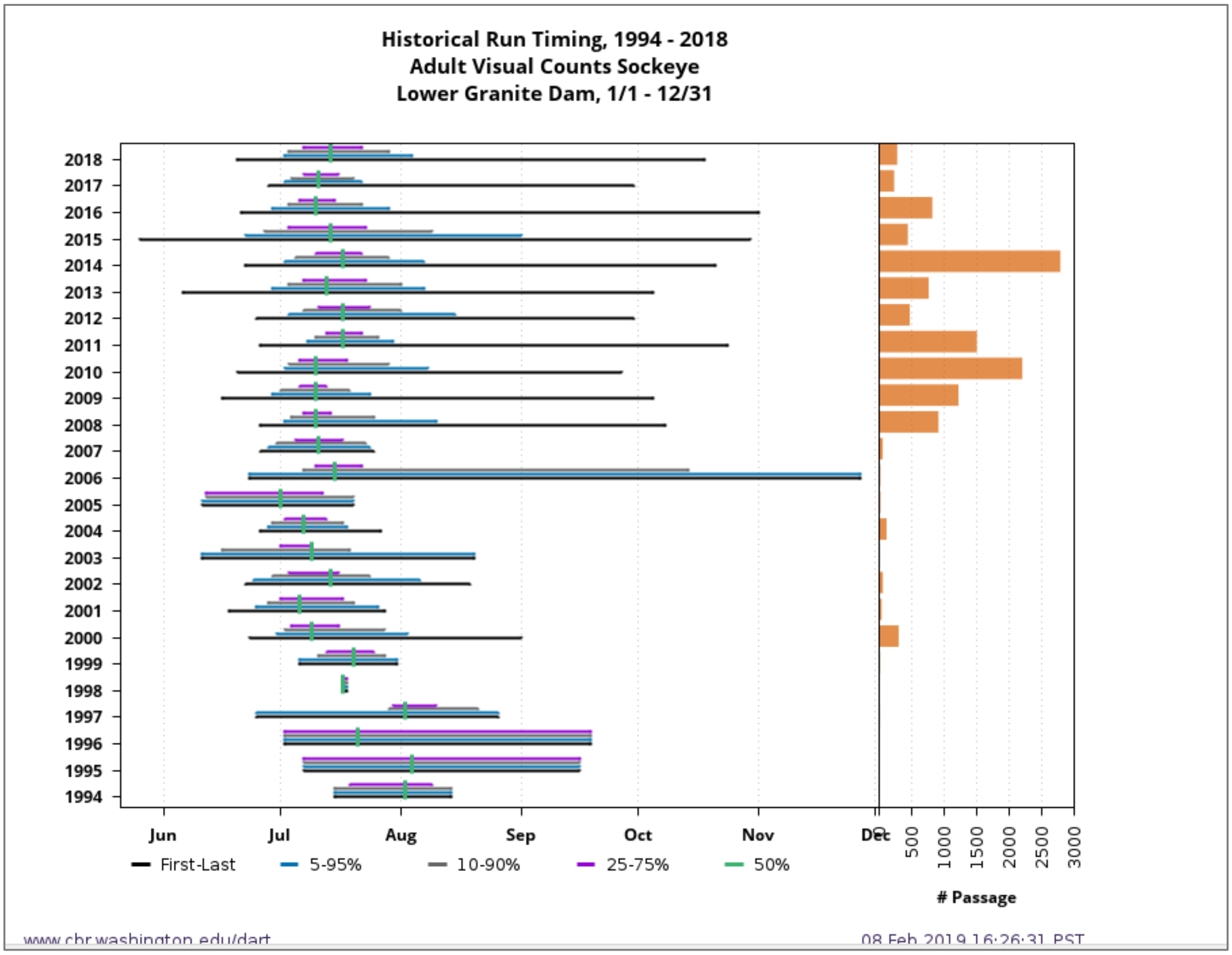

Figure 3.7. Abundance and timing of Snake River sockeye salmon passing Lower Granite Dam 1994-2018 (Source: University of Washington DART database. Accessed 2/8/19).

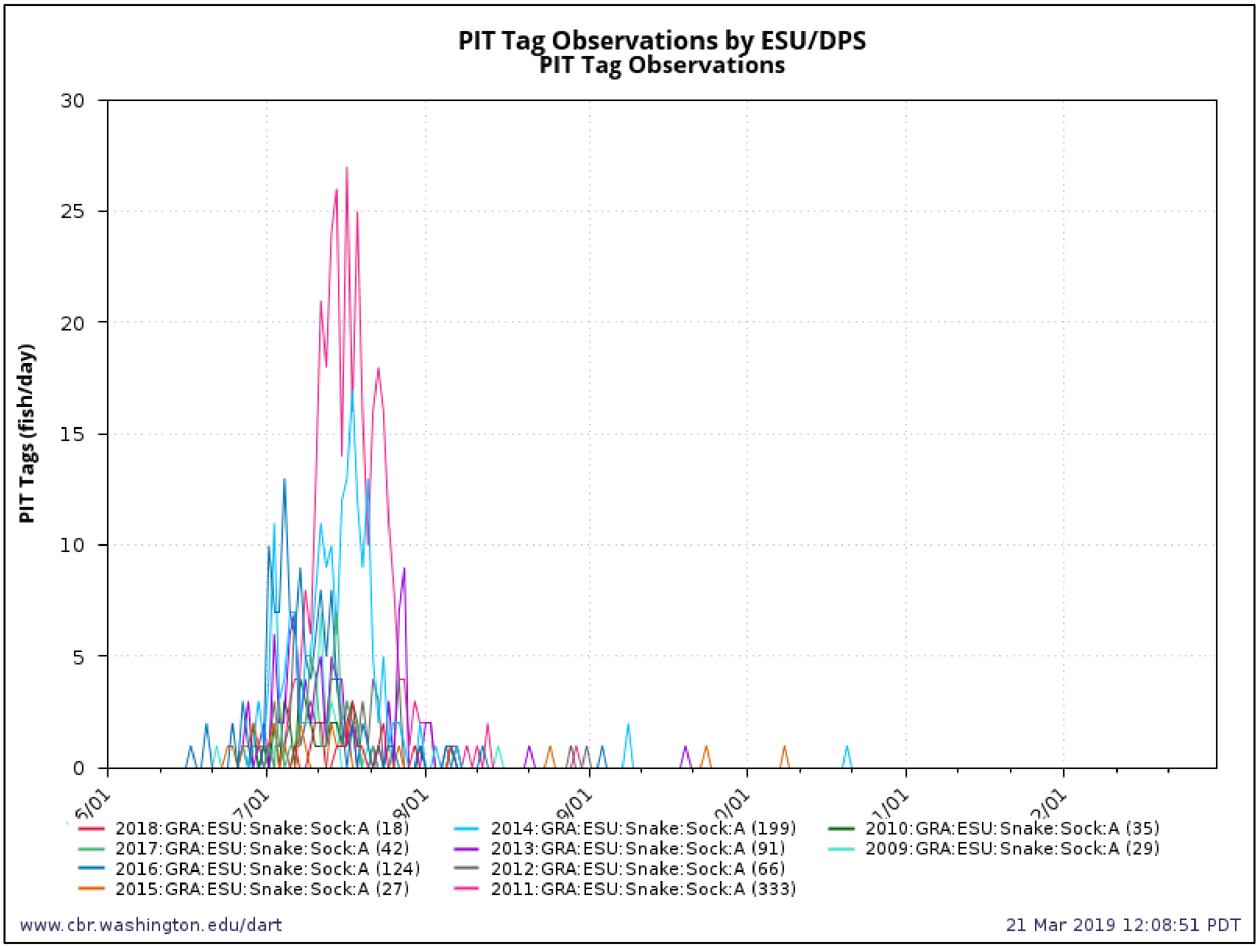

Figure 3.8. Pit-tag data for a subset of the Sockeye passing Lower Granite Dam (Source: University of Washington DART database).

3.7. Snake River Basin steelhead (*O. mykiss)*

The Snake River Basin (SRB) steelhead was listed as a threatened ESU on August 18, 1997 (62 FR 43937), with a revised listing as a Distinct Population Segment (DPS) on January 5, 2006 (71 FR 834). In 2016, NMFS conducted a 5-year review of the status of the species and based on the best available scientific information determined that the “threatened” classification remained appropriate (NMFS 2016; 81 FR 33468). This DPS includes all naturally spawning steelhead populations below natural and manmade impassable barriers in streams in the SRB. Six artificial propagation programs are considered part of the DPS: Tucannon River, Dworshak NFH, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha River Hatchery steelhead hatchery programs. NMFS has determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the DPS [71 FR 834]. The SRB steelhead listing does not include resident O. mykiss (rainbow trout) that co-occur with (migratory) steelhead. [62 FR 43937].

Two major genetic groups or "subspecies" of steelhead occur on the west coast of the United States: a coastal group and an inland group, separated on the Fraser and Columbia River basins by the Cascade Crest. Historically, steelhead likely inhabited most coastal streams in Washington, Oregon, and California, as well as many inland streams in these states and in Idaho. However, during the 20th century, over 23 indigenous, naturally reproducing stocks of steelhead are believed to have been extirpated, and many more are thought to be in decline in numerous coastal and inland streams.

The Snake River Basin steelhead Distinct Population Segment (DPS) occupies the SRB, which drains portions of southeastern Washington, northeastern Oregon, and north/central Idaho. The Snake River flows through terrain that is warmer and drier on an annual basis than the upper Columbia Basin or other drainages to the north. Geologically, the land forms are older and much more eroded than most other steelhead habitat. Collectively, the environmental factors of the Snake River Basin result in a river that is warmer and more turbid, with higher pH and alkalinity, than is found elsewhere in the range of inland steelhead.

In their 2003 document, NMFS identified steelhead-supporting tributaries to Hells Canyon as an unaffiliated population likely dependent historically on upstream populations (NMFS 2005). NMFS has now linked these small tributaries to likely spawning concentrations in Wildhorse Creek and Powder River. The Hells Canyon tributary region is now designated as a component of the Wildhorse-Powder population and belongs to the Hells Canyon genetically similar major population group (MPG). Remaining populations in the Hells Canyon MPG have been extirpated. Historically, this area was occupied but is currently inaccessible to steelhead populations. NMFS has designated 15 populations in four MPGs in areas currently blocked by the Hells Canyon dam complex and other upstream dams.

3.7.1. Life History

Steelhead exhibit one of the most complex life histories of any salmonid species. They can be anadromous or freshwater resident (and under some circumstances, yield offspring of the opposite form). Resident forms are usually referred to as "rainbow" or "redband" trout, while anadromous life forms are termed "steelhead." Two subspecies of steelhead with anadromous life history are recognized in North America. These are: *O. mykiss irideus* (the coastal subspecies), which includes coastal populations from Alaska to California (including the Sacramento River), and *O. mykiss gairdneri* (the inland subspecies), which includes populations from the interior Columbia, Snake, and Fraser rivers (Good et al. 2005).

Most steelhead adults migrate to their natal stream to spawn as 4- or 5-year-olds. Spawning occurs over coarse substrates (gravel) in cold, fast-flowing streams with highly oxygenated waters, and spawning may occur more than once (NMFS 2017). Depending on water temperature, steelhead eggs may incubate in redds for 1.5 to 4 months before hatching as alevins (larval stage dependent on yolk sac as food). Following yolk sac absorption, alevins emerge from the gravel as young juveniles (fry) and begin actively feeding.

Juvenile steelhead typically reside in freshwater streams for 1-4 years before migrating into estuaries to smoltify. Most rear for 2 years and some for as many as 7 years. A small number of steelhead return to freshwater after their first year only to migrate back out without spawning;

this behavior is irregular among salmonid species. The ocean phase lasts 2-3 years prior to adult return to the freshwater system. Steelhead typically feed on zooplankton as juveniles and shift to larger insects, mollusks, crustaceans, and fish as adults (NMFS 2017).

Fisheries managers have traditionally divided steelhead into two basic reproductive ecotypes, based on the state of sexual maturity at the time of river entry and duration of spawning migration. The stream-maturing type (summer-run steelhead in the Pacific Northwest and northern California) enter freshwater in a sexually immature condition between May and October and require several months to mature and spawn. The ocean-maturing type (winter-run steelhead in the Pacific Northwest and northern California) enter freshwater between November and April, with well-developed gonads, and spawn shortly thereafter. These two reproductive ecotypes are more commonly referred to by their season of freshwater entry "summer" and "winter" steelhead. Coastal streams are dominated by winter-run steelhead, whereas inland steelhead of the Columbia and Snake River basins are almost exclusively summer-run steelhead (Good et al. 2005).

Fish of the Snake River Basin steelhead ESU are summer steelhead, and comprise two groups, A- and B-run, based on migration timing, ocean-age, and adult size. A-run steelhead, thought to be predominately age-1-ocean, enter freshwater during June through August. B-run steelhead, thought to be age-2-ocean, enter freshwater during August through October. B-run steelhead typically are 75 to 100 mm longer at the same age. Both groups usually smolt as 2- or 3-year-olds (Busby et al. 1996, BPA 1992, Hassemer 1992).

As with Chinook, migrating steelhead make extensive use of cold-water areas (Keefer et al. 2018). Steelhead may spend prolonged periods in cold water areas (e.g. mouths of tributaries) because steelhead enter freshwater in pre-mature state and have a protracted period in freshwater before spawning. After holding over the winter in larger rivers in the SRB, steelhead disperse into smaller tributaries to access spawning habitat. Earlier dispersal occurs at lower elevations, and later dispersal occurs at higher elevations. SRB steelhead spawn the following spring from March through May. Their eggs incubate in redds for up to four months before hatching.

Juveniles emerge from the gravels four to eight weeks after hatching, and move into shallow, low-velocity areas in side channels and along channel margins, where they can escape high velocities and predators (Everest and Chapman 1972). Juvenile steelhead then progressively move toward deeper water as they grow (Bjornn and Rieser 1991). Juveniles typically reside in freshwater for one to three years. Smolts migrate downstream during spring runoff, which occurs from March to mid-June depending on elevation, and typically spend one to two years in the ocean.

3.7.2. Stressors and Threats

Hydrosystem projects create substantial habitat blockages in this ESU; the major ones are the Hells Canyon Dam complex (mainstem Snake River) and Dworshak Dam (North Fork Clearwater River). Minor blockages to fish passage are common in tributaries throughout the region. As with the other salmon species, steelhead have been affected by various human activities that have contributed to their decline. Spawning and rearing areas have been degraded by human land management practices including overgrazing, historical gold dredging and other practices that increase sedimentation. Impacts of climate change are also considered factors for decline (i.e. prolonged drought conditions). As reviewed in NMFS 2011b; NMFS 2011c, limiting factors include:

• Degradation of floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality
• Increased water temperature
• Harvest-related effects, particularly for B-run steelhead
• Predation
• Genetic diversity effects from out-of-population hatchery releases

3.7.3. Population Trend and Risk

Naturally produced fish make up only a small fraction of the total adult run of the Snake River steelhead ESU. Although several large production hatcheries exist throughout this ESU, relatively few data exist regarding the numbers and relative distribution of hatchery fish that spawn naturally, or the consequences of such spawning when they do occur. Significant increases in 2000 and 2001 in adult returns in some populations and evidence for high smolt-adult survival indicate that populations in this ESU are still capable of responding to favorable environmental conditions. Besides the recent increases, abundance in most populations for which there are adequate data are well below interim recovery targets.

For the entire SR steelhead ESU, NMFS (2000) estimates that the median population growth rate (lambda) over a base period from 1990 through 1998 ranges from 0.91 to 0.70, decreasing as the effectiveness of hatchery fish spawning in the wild increases compared with that of fish of wild origin (Tables B-2a and B-2b in McClure et al. 2000). The main producer of steelhead in the Columbia River Basin is the Snake River. In 2002, the turnoff into the Snake River was about 210,000, 71 percent of the total counted at McNary Dam (286,805). The 2002 Snake River steelhead count was about twice the 10-year average. The numbers of wild steelhead (nonclipped adipose fin) increased to a mean of 55,000 in the Snake River in 2002 (FPC 2003).

The NMFS included the following summary in their 2004 biological opinion for Consultation on Remand for Operation of the Columbia River Power System (NMFS 2004): ‘The lack of information on adult spawning escapement to many tributary production areas makes it difficult to quantitatively assess the viability of the SR steelhead ESU. Estimates of annual returns are limited to estimates of aggregate numbers over Lower Granite Dam and spawner estimates for the Tucannon, Grande Ronde, and Imnaha rivers. In their preliminary report, Fisher and Hinrichsen (2004) estimated that the geometric mean of the natural-origin run was 37,784 during 2001-2003, a 253% increase over the 1996-2000 period (10,694 steelhead). The slope of the population trend increased 9.3% (from 1.00 to 1.10) when the counts for 2001-2003 were added to the 1990-2000 data series. These data indicate that, at least in the short term, the natural-origin run has been increasing’.

NOAA states in the 2015 Opinion on USEPA’s Action on OR WQS that the status of most populations in this DPS remains highly uncertain. Population-level natural origin abundance and productivity inferred from aggregate data and juvenile indices indicate that many populations are below the minimum combinations defined by the IC-TRT viability criteria.

3.7.4. Critical Habitat

Critical habitat was designated by the NMFS on September 2, 2005 (70 FR 52630). Critical habitat for the steelhead consists of river reaches of the Columbia, Snake, and Salmon Rivers, and all tributaries of the Snake and Salmon River presently or historically accessible to Snake River steelhead (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dam). This designation includes including Clearwater, Grande Ronde, Selway and Tucannon Rivers. PBFs for the Snake River Basin DPS of steelhead are compiled in Table 3.3.

3.7.5. Time of Use of the Action Area

Steelhead of the Snake River DPS are considered summer-run steelhead due to their adult migration pattern. These fish typically enter the Columbia River from May to October. In the Snake River, most passing Lower Granite Dam in mid-September through late October (Figure 3.9 and Figure 3.10). After holding over the winter in mainstem reaches of the Columbia River or Snake River or in reservoirs, summer-run steelhead spawn 6 to 11 months following initiation of migration. Spawning occurs in tributaries the following spring (typically from March to June) (Busby-Status Review 1996).

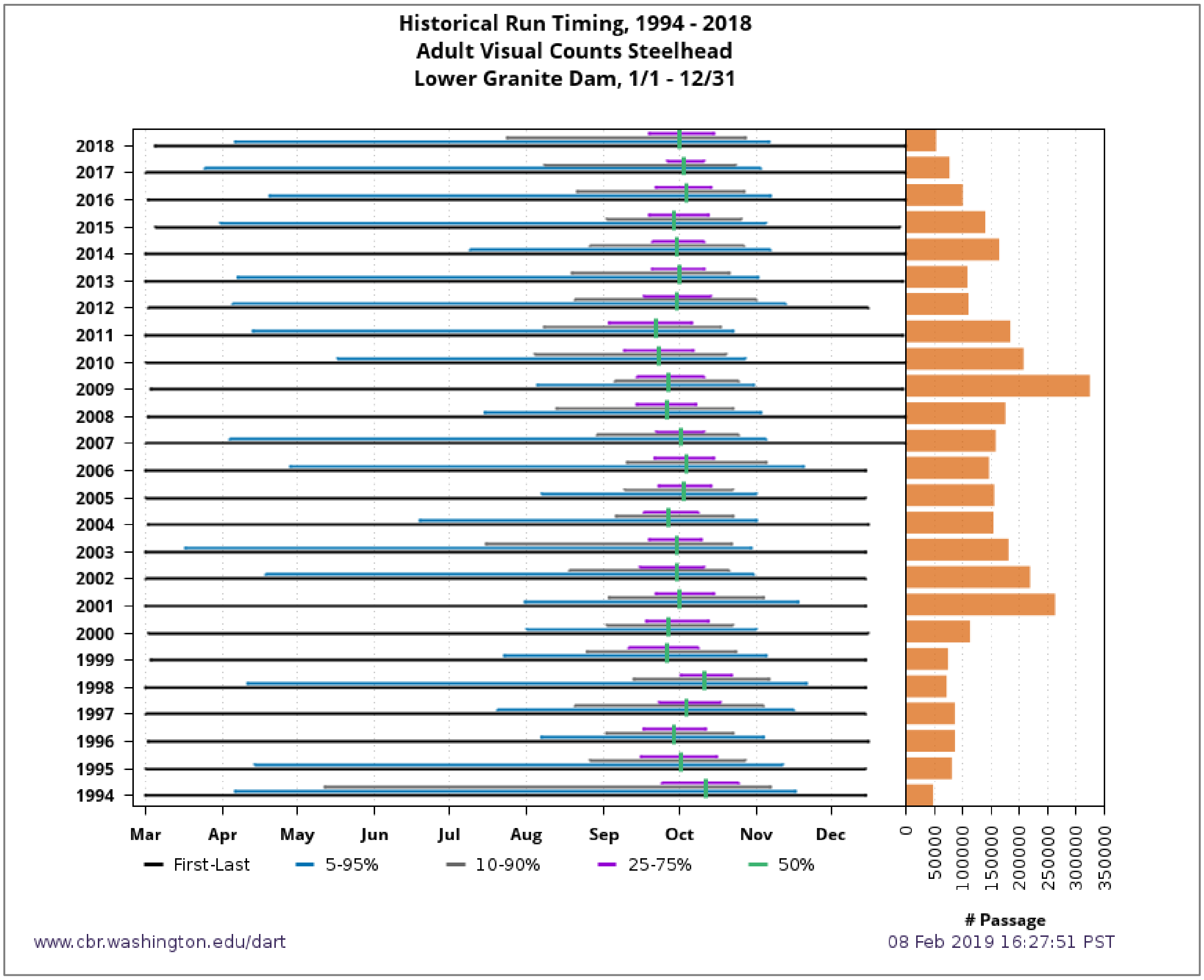

Figure 3.9. Abundance and timing of Snake River steelhead passing Lower Granite Dam 1994-2018 (Source: University of Washington DART database. Accessed 2/8/19).

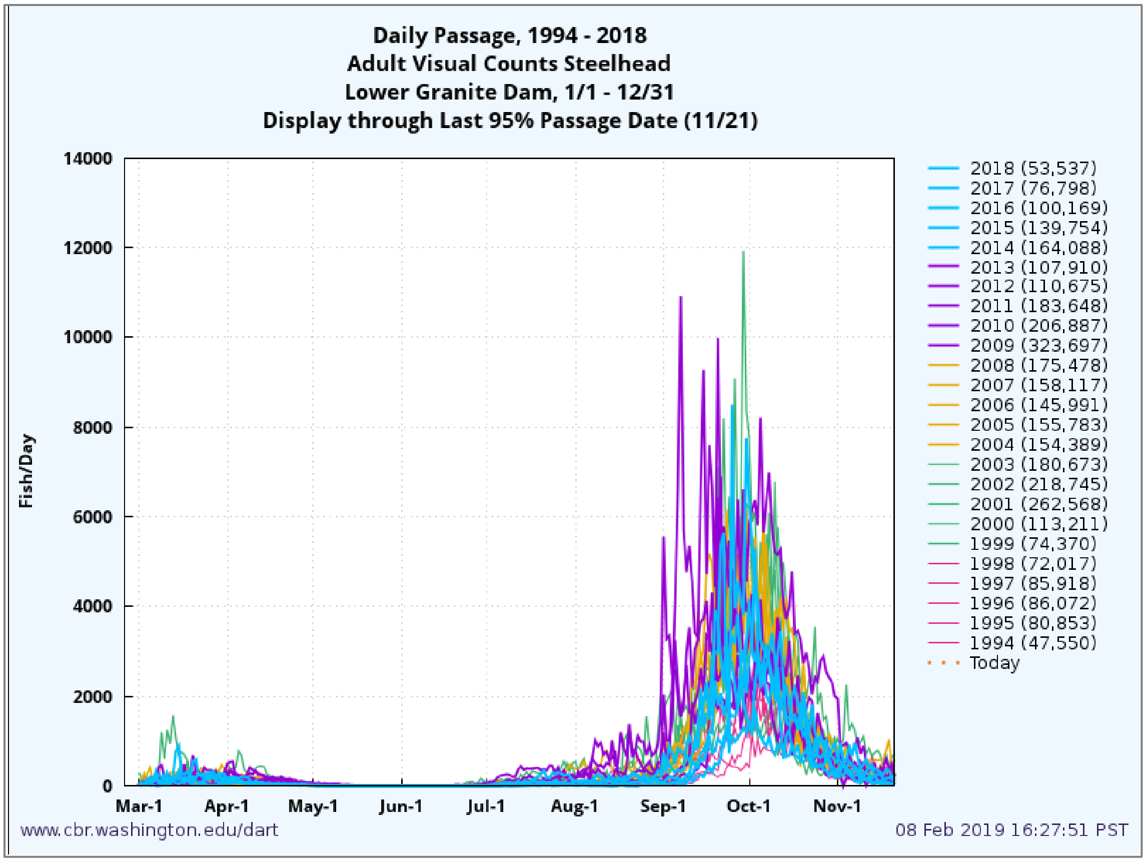

Figure 3.10. Adult steelhead passage counts at Lower Granite dam by year (Source: University of Washington DART data base. Accessed February 8th, 2019).

3.8. Bull trout—*S. confluentus*

The coterminous US population of bull trout was listed as threatened on November 1, 1999 (64 FR 58910). Within the area covered by this listing, bull trout are known to occur in Idaho, Montana, Nevada, Oregon, and Washington. Bull trout are members of the Salvelinus genus of salmoninae family. This genus referred to as 'char' also includes lake trout, arctic char, and brook trout.

3.8.1. Distribution

Bull trout range from Puget Sound throughout the Columbia River and Snake River basins, extending east to headwater streams in Montana and Idaho, and into Canada. Bull trout occur in the Klamath River Basin of south-central Oregon; the Jarbidge River in Nevada; the Willamette River Basin in Oregon; Pacific Coast drainages of Washington, including Puget Sound; major rivers within the Columbia River Basin in Idaho, Oregon, Washington, and Montana; and the St. Mary-Belly River, east of the Continental Divide in northwestern Montana (Bond 1992; Brewin and Brewin 1997; Cavender 1978; Leary and Allendorf 1997). The Columbia River population segment comprises 386 bull trout populations in Idaho, Montana, Oregon, and Washington, with additional populations in British Columbia. The Columbia River population segment includes the entire Columbia River Basin and all its tributaries, excluding the isolated bull trout populations found in the Jarbridge River in Nevada.

3.8.2. Life History

Bull trout exhibit both resident and migratory life history strategies. Both resident and migratory forms may be found together, and either form may produce offspring exhibiting either resident or migratory behavior (Rieman and McIntyre 1993). Resident fish live their whole lives near areas where they were spawned. In smaller, high elevation streams these fish seldom reach size of over 30 cm (Brown 1994; USFWS 2002). In contrast migratory bull trout grow to larger size and may migrate long distances to access various habitats for spawning, overwintering, or rearing(Fraley and Shepard 1989; Goetz 1989). Three general forms of migratory bull trout are recognized: 1) fluvial fish are spawned in small headwater streams and then migrate to larger streams or rivers where they mature, 2) adfluvial fish spawned in small streams but migrate to lakes or reservoirs to grow and mature , and) anadromous fish spawned in freshwater and migrate to salt water where they grow to maturity(Cavender 1978; McPhail and Baxter 1996; WDFW et al. 1997)..

Bull trout typically spawn from August through November after temperatures drop below 9°C, during periods of increasing flows and decreasing water temperatures. Spawners use streams with abundant cold, unpolluted water, clean gravel and cobble substrate, and gentle stream slopes. Many spawning areas are associated with cold water springs or areas where stream flow is influenced by ground water (Baxter et al. 1997; Pratt 1992; Rieman and McIntyre 1993; Rieman et al. 1997).

Bull trout eggs incubate from 100 to 145 days, usually in winter, after which the alevins require 65 to 90 days to absorb their yolk sacs (Buchanan et al. 1997). They remain in redds as fry for up to 3 weeks before emergence in early April through May, depending on water temperatures and increasing stream flows (Ratliff and Howell 1992 in Howell and Buchanan 1992; Pratt 1992). Bull trout reach sexual maturity at 4 to 7 years of age and are known to live up to 12 years. They are iteroparous, meaning that they may spawn more than once in a lifetime.

Bull trout are opportunistic feeders, with food habits primarily a function of size and life-history strategy. Resident and juvenile migratory bull trout eat terrestrial and aquatic insects but shift to preying on other fish as they grow larger. Subadult and adult migratory bull trout feed on various fish species (Brown 1994; Donald and Alger 1993; Fraley and Shepard 1989; Leathe and Graham 1982).

Bull trout are primarily found in colder streams (below 15°C; 59°F) (Fraley and Shepard 1989; Pratt 1992; Rieman and McIntyre 1993), though they may be found in warmer waters that have access to colder refuges. Bull trout are seldom found in waters where temperatures are warmer than 15°C to 18°C. Besides very cold water, bull trout require stable stream channels, clean spawning gravel, complex and diverse cover, and unblocked migration routes (USFWS 2002). Early life stages of bull trout, specifically the developing embryo, require the highest inter-gravel dissolved oxygen levels, and are the most sensitive life stage to reduced oxygen levels. The oxygen demand of embryos depends on temperature and stage of development, with the greatest dissolved oxygen required just prior to hatching.

3.8.3. Stressors and Threats

Bull trout are vulnerable to many of the same threats that have reduced salmon populations. Throughout their range, bull trout are threatened by the combined effects of habitat degradation, fragmentation, and alterations associated with dewatering, road construction and maintenance, mining, grazing, the blockage of migratory corridors by dams or other diversion structures, entrainment in diversion channels, and introduced non-native species (64 FR 58910).

-- Because of their need for very cold waters and long incubation time, bull trout are sensitive to the land management activities that have resulted in increased water temperatures, poor water quality, and degraded stream habitat, especially along larger river systems and streams located in valley bottoms.

--Degraded conditions have severely reduced or eliminated migratory bull trout as water temperature, stream flow, and other water quality parameters fall below the range of conditions that these fish can tolerate. In many watersheds, remaining bull trout are smaller, resident fish isolated in headwater streams.

--Brook trout, introduced throughout much of the range of bull trout, easily hybridize with them, producing sterile offspring. Brook trout also reproduce earlier and at a higher rate than bull trout, so bull trout populations are often supplanted by these nonnatives. Also, competition with non-native brown trout, lake trout, and brook trout can be detrimental to bull trout populations

--Dams and other instream structures affect bull trout by blocking migration routes, altering water temperatures, and killing fish as they pass through and over dams or are trapped in irrigation and other diversion structures (USFWS 2002).

--bull trout are especially vulnerable to the impacts of climate change given that spawning and rearing are constrained by their location in upper watersheds and the requirement for cold water temperatures (Battin et al. 2007; Rieman et al. 2007).

--Additional threats to bull trout include industrial development and urbanization, timber harvest, and poaching or bycatch

3.8.4. Population Trend and Risk

Bull trout populations within the Columbia River population segment have declined from historic levels and are generally considered to be isolated and remnant. In Idaho, bull trout were historically found in the major tributaries in the Columbia and Snake Rivers. Currently, most bull trout populations are confined to headwater areas of tributaries to the Columbia, Snake, and Klamath rivers. While bull trout occur over a large area, their distribution has contracted, and abundance has declined. Several local extinctions have been documented. Many of the remaining populations are small and isolated from each other, making them more susceptible to local extinctions.

3.8.5. Critical Habitat

A final ruling on critical habitat for bull trout in the coterminous US was made on October 18, 2010 (effective November 17, 2010) (75 FR 63898). Critical habitat for bull trout includes approximately 32,187 km (20,000 miles) of riverine habitat, 1,207 km (750 miles) of marine shoreline, and 197,487 ha (488,001 acres) of lacustrine habitat. Critical habitat spans Washington, Oregon, Idaho, Nevada, and Montana.

In Idaho, designated critical habitat for the bull trout includes areas of 24 counties, including over 8,771 stream miles, and over 170,000 lake and reservoir acres. Critical Habitat Units in Idaho include the Imnaha River Basin, Sheep and Granite Creeks, Powder River Basin, Hells Canyon Complex, Clearwater River, Mainstem Upper Columbia River, Mainstem Snake River, Malheur River Basin, Jarbidge River, Southwest Idaho Basins, Salmon River, Little Lost River, Coeur d'Alene River Basin, Kootenai River Basin, Clark Fork River Basin, and the St. Mary River Basin.

The PBFs determined to be essential to the conservation of bull trout are:

1. Springs, seeps, groundwater sources, and subsurface water connectivity (hyporheic flows) to contribute to water quality and quantity and provide thermal refugia;

2. Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including but not limited to permanent, partial, intermittent, or seasonal barriers;

3. An abundance of food, including terrestrial organisms of riparian origin, aquatic macroinvertebrates, and forage fish;

4. Complex shorelines with features such as large wood, side channels, pools, undercut banks, and unembedded substrates, to provide a variety of depths, gradients, velocities, and structure;

5.Water temperatures ranging from 2 to 15°C (36 to 59°F), with adequate thermal refugia available for temperatures that exceed the upper end of this range;

6. In spawning and rearing areas, substrate of sufficient amount, size, and composition to ensure success of egg and embryo overwinter survival, fry emergence, and young-of-the-year and juvenile survival. A minimal amount of fine sediment, generally ranging in size from silt to coarse sand, embedded in larger substrates, is characteristic of these conditions. The size and amounts of fine sediment suitable to bull trout will likely vary from system to system.

7. A natural hydrograph, including peak, high, low, and base flows within historic and seasonal ranges or, if flows are controlled, minimal flow departures from a natural hydrograph.

8. Sufficient water quality and quantity to sustain normal reproduction, growth, and survival; and

9. Low occurrence of nonnative predatory (e.g., lake trout, walleye, northern pike, smallmouth bass), interbreeding (e.g., brook trout), or competing (e.g., brown trout) species.

3.8.6. Period of Presence in the Action Area

As described in the final Recovery Plan (USFWS 2015), the Action Area is within the USFWS designated bull trout Mid-Columbia Recovery Unit and in the geographic region within this recovery unit called the Lower Snake River geographic region. This includes 11 core areas that flow into the Snake River between its confluence with the Columbia River and the Hells Canyon Dam (i.e., Clearwater, Tucannon, Asotin, Grande Ronde, and the Imnaha basins).

According to ODFW (Pers. Comm. Kyle Bratcher, ODFW February 2019 to R. Labiosa USEPA), the 'vast majority of bull trout in the Snake River originate in the Imnaha River'. Based on PIT tag data 2011-2014 (ODFW unpublished data supplied by Kyle Bratcher), bull trout migrate upstream into the Imnaha from the Snake beginning in the spring and continuing into the summer months (Table 3.6). The average period the upstream movement into the Imnaha is in late April based on these four years of data. Adults move out of the Imnaha River in the fall with the average period of late November. The following graph (Figure 3.11) illustrates the temporal distribution of these upstream and downstream movements by adult bull trout. In general, the total counts are low and time-periods are protracted relative to the number of bull trout detected. These data indicate bull trout have moved out of the mainstem Snake during late spring through mid- fall.

Table 3.6. Bull trout upstream and downstream passage dates detected at lower Imnaha River (Source: ODFW unpublished data from Kyle Bratcher).

Imnaha River #1 (IR1) (Rkm 7)		
Mean Migration Date	**Date Range**	**n**
Upstream (SPRING)		
4/19/2011	3/10/2011 - 7/10/2011	36
4/22/2012	2/25/2012 - 7/8/2012	95
4/29/2013	2/17/2013 - 8/18/2013	157
4/21/2014	2/11/2014 - 6/28/2014	172
Downstream (FALL)		
11/25/2011	10/28/2011 - 12/28/2011	30
11/22/2012	10/7/2012 - 1/18/2013	35
11/19/2013	10/9/2013 - 1/11/2014	57
11/25/2014	9/28/2014 - 1/27/2015	62

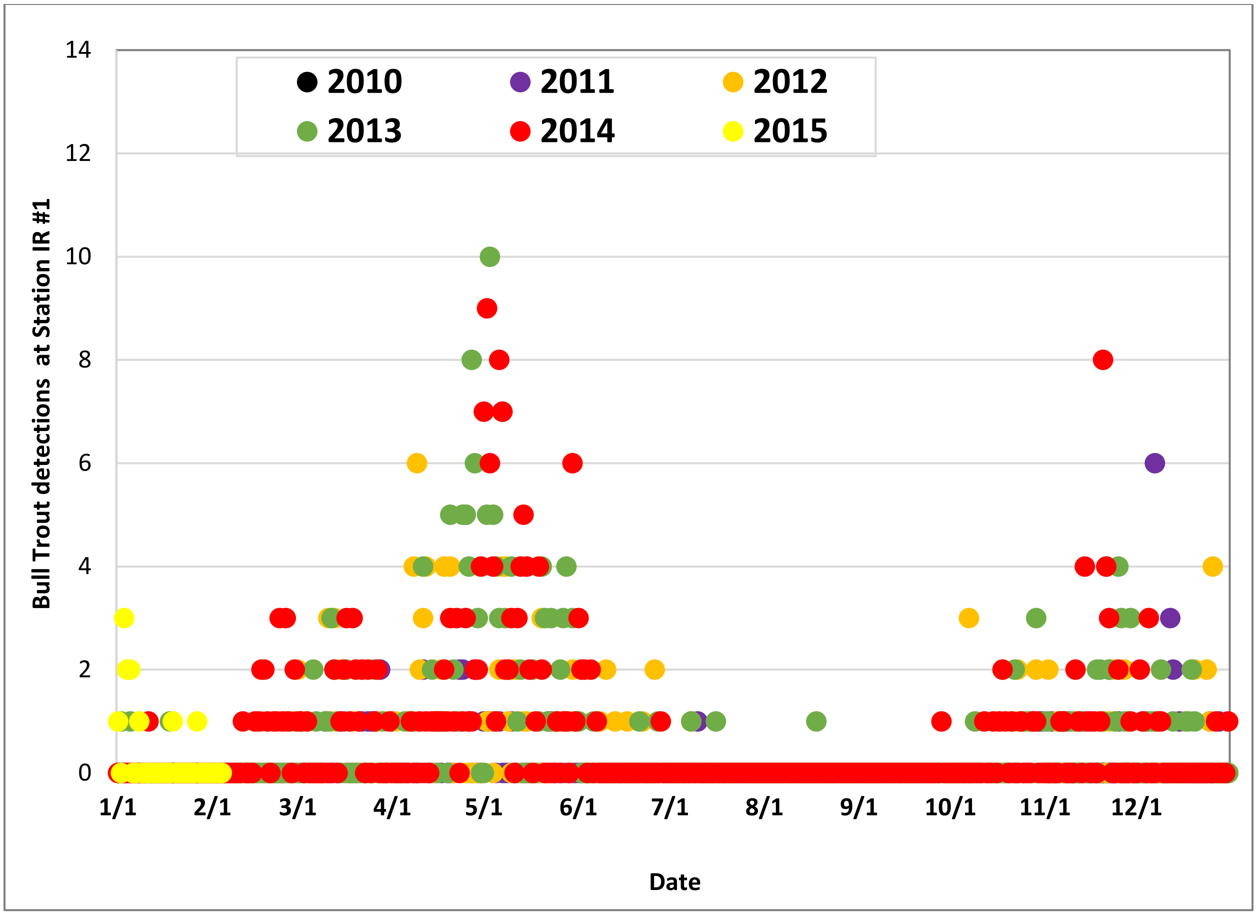

Figure 3.11. Detections of PIT tagged bull trout movements in spring-summer (upstream) and fall (downstream) at the Lower Imnaha station (RKm 7), 2010-2015 (Source: Idaho Power, unpublished data from Rick Wilkerson; Summarized in memo by P. Leinenbach, Appendix C).

3.9. Southern Resident killer whale (*Orcinus orca)*

The distinct population segment (DPS) of Southern Resident killer whales (Orcinus orca) was listed as endangered under the ESA on November 18, 2005 (70 FR 69903). Southern Residents are designated as “depleted” and “strategic” under the Marine Mammal Protection Act (MMPA) (68 FR 31980, May 29, 2003). NMFS issued the final recovery plan for Southern Residents in January 2008 (NMFS 2008a).

Killer whales (often referred to as “orcas”) are the largest odontocete (toothed) dolphin species; adults tend to be 6.1 to 7.3 m (20 to 24 ft) in length, but killer whales may grow as large as 9.8 m (32 ft) (NOAA Fisheries 2017a). SRKWs live in inland and coastal marine waters, generally 49 to 55 m (160 to 180 ft) deep (Noren and Hauser 2016). Southern Residents are highly mobile and can travel up to 86 miles in a single day (Erickson 1978; Baird 2000).

Killer whales of related matrilines group together to form pods. Three pods – J, K, and L – make up the Southern Resident community. Clans are composed of pods with similar vocal dialects and all three pods of the Southern Residents are part of J clan. The follow description of the Southern Resident Killer Whale (SRKW) summarizes information taken largely from the recovery plan and recent 5-year status review (NMFS 2011d), as well as more recent information.

3.9.1. Range and Distribution

SRKWs occur throughout the coastal waters of Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as southeast Alaska. They are present in the Salish Sea (Puget Sound, Strait of Juan de Fuca, and the Strait of Georgia) from spring to fall each year (NOAA Fisheries 2017a). In winter, some SRKWs remain in the Salish Sea, while others travel along the Washington, Oregon, and California coasts (as far south as central California) (NWFSC 2015). SRKWs may also travel north along the British Columbia border as far as the Queen Charlotte Islands and southeast Alaska. Between late spring and early autumn, SRKWs spend a significant portion of time in the Georgia Basin (Canada) and around the San Juan Islands of Washington following incoming salmon runs (NOAA Fisheries 2017a). Satellite- tagged animals and tracking has identified an important winter through spring foraging area along the west coast of Washington down to the mouth of the Columbia River (Hanson et al. 2013). Although SRKWs can occur along the outer coast of Washington and Oregon at any time of the year, occurrence along the outer coast is more likely from late autumn to early spring.

SRKWs co-exist in areas with West Coast transient killer whales, but resident and transient groups generally do not have significant interactions (e.g., socializing or attacking one another) (Barrett-Lennard and Heise 2007).

3.9.2. Life History

3.9.2.1. Reproduction

Southern Resident killer whales are a long-lived species with late onset of sexual maturity (review in NMFS 2008a). Females produce a low number of surviving calves over the course of their reproductive life span (Bain 1990; Olesiuk et al. 1990). The average interbirth interval for reproductive Southern Resident females is 6.1 years (Olesiuk et al. 2005). Mothers and offspring maintain highly stable social bonds throughout their lives, which is the basis for the matrilineal social structure in the Southern Resident population (Baird 2000; Bigg et al. 1990; Ford et al. 2000).

3.9.2.2. Social groups

The familial pods include 20 to 40 individuals led by a dominant matriarch (NOAA Fisheries 2017b, a). Stable social groups tend to include 2 to 15 individuals at a time, but large, temporary aggregations of the entire population occur, particularly in the summer (NOAA Fisheries 2017a). Aggregation and separation of groups tends to follow seasonal trends in prey availability and courtship and mating activities. Temporary associations of the pods, called "superpods," of 50 or more individuals may form for a matter of days during late summer, consistent with when whales are mating (Barrett-Lennard and Heise 2007).

Other social types are transient killer whales and offshore killer whales. Transient killer whales generally travel in small groups and will hunt marine mammals. Offshore killer whales are uncommon, although groups of over 100 have been observed.

3.9.2.3. Behavior

Observations of Southern Resident Killer Whale (SRKW) behavior indicates that their active time is primarily budgeted to travel (70.4%), followed by foraging (21%), rest (6.8%), and socialization (1.8%) (Noren and Hauser 2016). Others have suggested that foraging accounts for a greater amount of activity, 40 to 67% (Ford and Ellis 2006). Diving tends to be concentrated within the upper 30 m (98 ft) of the water column, with deeper dives of 100 to 200 m (328 to 656 ft) (or more) being occasional (Baird et al. 2005). Diving activity is greatest during the day, and dive depths and frequencies are greater for males than females (in adults) but are not greater for adults than juveniles (on average) (Baird et al. 2005). Killer whales are relatively recognizable due to their distinctive coloring and high level of surface activity (e.g., breaching and tail slapping), though SRKWs cannot easily be differentiated from transient individuals. Killer whales communicate, navigate, and hunt using several types of calls, whistles, and clicks (NOAA Fisheries 2017b).

3.9.2.4. Diet

Salmon are identified as their primary prey of SRKW with a high percentage consumed during spring, summer and fall (Ford and Ellis 2006; Hanson et al. 2010c). Feeding records for Southern and Northern Residents show a predominant consumption of Chinook salmon during late spring to fall (Ford and Ellis 2006). Chum salmon are also taken in significant amounts, especially in fall. Other salmon eaten include coho, pink, steelhead, and sockeye. They also consume non-salmonid fishes included Pacific herring, sablefish, Pacific halibut, quillback and yelloweye rockfish (*Sebastes maliger*), lingcod (*Ophiodon elongates*), and Dover sole (*Microstomus pacificus*) (Ford et al. 1998; Hanson et al. 2010c) and squid.

SRKWs preferentially consume Chinook salmon. Chinook salmon were the primary prey despite the much lower abundance of Chinook salmon in the study area in comparison to other salmonid fishes (primarily sockeye salmon). Though mechanisms are not well known, factors of potential importance include the species' large size, high fat and energy content, and year-round occurrence in the area. Killer whales also captured older (i.e., larger) than average Chinook salmon (Ford and Ellis 2006). Recent research suggests that killer whales are capable of detecting, localizing and recognizing Chinook salmon through their ability to distinguish Chinook salmon echo structure as different from other salmon (Au et al. 2010).

Scale and tissue sampling in inland waters from May to September indicate that the Southern Residents' diet consists of a high percentage of Chinook salmon, with an overall average of 88% Chinook across the timeframe and monthly proportions as high as >90% Chinook salmon (i.e., July: 98% and August: 92%, Hanson et al. 2010c). The significance of the dominance of Chinook in the diet of SRKW is discussed at the end of this section.

3.9.2.5. Movement

Based on acoustic activity of whales, it is inferred that whale movements and presence are driven by local availability and abundance of salmon (Hanson et al. 2013), suggesting that the prey base is the most important habitat element for SRKWs. Recent evidence shows that K and L pods are spending significantly more time off of the Columbia River in March than was previously recognized, suggesting the importance of Columbia River spring Chinook salmon in their diet (Hanson et al. 2013).

Late summer and early fall movements of Southern Residents in the Georgia Basin are consistent, with strong site fidelity shown to the region generally and high occurrence in the San Juan Island area (Hanson and Emmons 2010; Hauser et al. 2007). There is inter-annual variability in arrival time and days present in inland waters from spring through fall, with late arrivals and fewer days present during spring in recent years potentially related to weak returns of spring and early summer Chinook salmon to the Fraser River (Hanson and Emmons 2010). Similarly, recent high occurrence in late summer may relate to greater than average Chinook salmon returns to South Thompson tributary of the Fraser River (Hanson and Emmons 2010). During fall and early winter, Southern Resident pods, (J pod in particular) expand their routine movements into Puget Sound, likely to take advantage of chum and Chinook salmon runs (Hanson et al. 2010a, Osborne 1999).

3.9.3. Stressors and Threats

Because of this population's small abundance, it is susceptible to demographic stochasticity — randomness in the pattern of births and deaths among individuals in a population. This can contribute to variance in a population's growth and extinction risk. Small populations are also vulnerable to environmental fluctuations that drive fluctuations in birth and death rates. Finally, small populations can have more vulnerability to variation in birth or death rates of individuals because of differences in their individual fitness.

Several factors identified in the final recovery plan for Southern Residents may be limiting recovery. It is likely that multiple threats are acting in concert to impact the whales. Although it is not clear which threat or threats are most significant to the survival and recovery of Southern Residents, all of the threats identified are potential limiting factors in their population dynamics (NMFS 2008a). Factors of concern include the follow:

Water quality: Water quality in areas inhabited by SRKW. Elevated concentrations of pollutants/contaminants in the Salish Sea and elsewhere have been linked to elevated concentrations in salmon and in killer whales (Krahn et al. 2007; Krahn et al. 2009; Lachmuth et al. 2010; Hickie et al. 2007). Once in the environment, many contaminants accumulate in biological tissues, and some biomagnify up the food chain, reaching high levels in long-lived apex predators like SRKWs. Maternal transfer of persistent and bioaccumulative contaminants from mother to offspring increases killer whale body burdens in subsequent generations (by increasing the baseline burden at birth) (Krahn et al. 2009). Elevated concentrations of pollutants may result in reduced immune function and/or reproductive capability and mortality (Krahn et al. 2007; Krahn et al. 2009).

Reduced quality and quantity of prey: Human influences have had profound impacts on the abundance of many prey species in the northeastern Pacific during the past 150 years, including salmon. As presented in the sections on threats to salmon, the health and abundance of wild salmon stocks have been negatively affected by altered or degraded freshwater and estuarine habitat, including numerous land use activities, from hydropower systems to urbanization, forestry, agriculture and development, harmful artificial propagation practices, and overfishing. Finally, the consequences of climate change contribute to habitat alterations (both in fresh and marine environments) that may negatively impact salmon and reduce productivity. The availability of adult salmon may be reduced in years following unfavorable conditions to the early life-stage growth and survival of salmon.

Nutritional stress: Another consequence of a reduced pry base is energy expenditure. When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources and as a chronic condition can lead to reduced body size and condition of individuals and lower reproductive and survival rates of a population (e.g., Trites and Donnelly 2003).

Other human activities: fishing, and disturbance caused vessels and noise pollution (e.g., caused by military activities) and risk from oil spills (NMFS 2008a) all pose possible risks to SRKWs.

3.9.4. Population Trend and Risk

The historical minimum abundance of SRKWs is estimated to 140 (Krahn et al. 2004; Olesiuk et al. 1990). Data are insufficient to estimate an upper bound but Several lines of evidence (i.e., known kills and removals [Olesiuk et al. 1990], salmon declines, and genetics [Krahn et al. 2004; Ford et al. 2011a]) all indicate that the population used to be much larger than it is now. A reasonable assumption is 400 as an upper bound (Krahn et al. 2004).

As of July 1, 2015, there were 81 RSKW (27 whales in J pod, 19 whales in K pod, and 35 whales in L pod). As of December 31, 2016, there were a total of 78 whales (CWR 2016). Of the three pods, the L pod is the largest at 35 members followed by J, which has 24 members, and then K, which only has 19 members (CWR 2016). The most recent count as of late 2018 is 74 whales. The estimated effective size of the population (based on the number of breeders under ideal genetic conditions) is very small at approximately 26 whales, or roughly 1/3 of the current population size (Ford et al. 2011a).

There are several demographic factors of the Southern Resident population that are cause for concern, namely the small number of breeding males (particularly in J and K pods), reduced fecundity, sub-adult survivorship in L pod, and the total number of individuals in the population (review in NMFS 2008a).

At present, the Southern Resident population has declined to essentially the same size that was estimated during the early 1960s, when it was considered likely to be depleted (Olesiuk et al. 1990). The population suffered an almost 20% decline from 1996-2001 (from 97 whales in 1996 to 81 whales in 2001), largely driven by lower survival rates in L pod. Since then, the overall population has fluctuated but still remained fairly consistent from 2002 to present (from 83 whales in 2002 to 81 whales on July 1, 2015). Over a recent 32-year period (1983-2014), population growth has been variable, with an average annual population growth rate of 0.1% and standard deviation of ± 3.2%.

Extinction Risk: In conjunction with the 2004 status review, NMFS conducted a population viability analysis (PVA) for Southern Resident killer whales (Krahn et al. 2004). Demographic information from the 1970s to fairly recently (1974-2003, 1990-2003, and 1994-2003) were considered to estimate extinction and quasi-extinction risk. We defined "quasi-extinction" as the stage at which 10 or fewer males or females remained, a threshold from which the population was not expected to recover. The analysis indicated that the Southern Resident killer whales have a range of extinction risk from 0.1 to 18.7% in 100 years and 1.9 to 94.2% in 300 years, and a range of quasi-extinction risk from 1 to 66.5% in 100 years and 3.6 to 98.3% in 300 years.

3.9.5. Critical Habitat

Approximately 6,630 sq km (2,560 square miles) of critical habitat were designated for the SRKW at the end of 2006 (71 FR 69054). This includes all US waters within the Salish Sea, excluding 18 areas designated for military use (291 sq km; 112 square miles), any waters less than 6.1 m (20 ft) deep (at extreme high tide), and Hood Canal. Military installations were excluded from critical habitat as a matter of national security. The critical habitat was subdivided into three areas that provide necessary habitat elements: a core summer area (Haro Strait and San Juan Islands), Puget Sound, and the Strait of Juan de Fuca. These subareas correspond with seasonal prey (e.g., salmon) concentrations.

The NMFS announced a 12-month finding on a petition from the Center for Biological Diversity to revise the critical habitat designation for the Southern Resident killer whale (Orcinus orca) Distinct Population Segment (DPS) (February 2014, 80 FR 9682). In February 2015, NMFS announced their intention to proceed with revisions to critical habitat. Although projected to have this complete by 2017, no new rules have been announced (Fed Reg. checked 2/2/19).

PBFs for this critical habitat are stated in 71 FR 69054 as: water quality to support growth and development; prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth; and passage conditions to allow for migrating, resting, and foraging.

3.9.6. Chinook as Prey for SRKW

In the BE we include a review of the quantity and quality of prey in the action area, because this is the relevance of the HCC action area for SRKW—<u>the quantity and quality of the prey that could be affected by the proposed action</u>. As demonstrated in the above sections, Chinook salmon are important in the diet of SRKW. Two Snake River Chinook stocks, Snake River fall Chinook and Snake River spring-summer Chinook salmon, appear on the list developed by NOAA Fisheries and the Washington Department of Fish and Wildlife regarding West Coast Chinook salmon stocks most important to Southern Resident killer whales (SRKW) (NOAA and WDFW 2018).

As stated in the life history section, Killer whales, including both the Southern Residents and other populations in Canada and Alaska, are large consumers of West Coast Chinook salmon in terms of biomass and numbers of adult Chinook salmon. The 74 Southern Resident killer whales, a small subset of all killer whales on the West Coast, consume an estimated 190,000 to 260,000 adult Chinook salmon each year. The Southern Residents depend on a diversity of salmon stocks that together provide the food they need throughout the year. The more diverse and healthy stocks available to the whales, the better they can withstand variable ocean conditions, climate change, and other factors that may affect the availability of salmon. (NMFS 2018b).

Given their more coastal distribution in fall, winter, and spring, K and L Pods are most likely to prey on Columbia/Snake River stocks directly. They would encounter Snake River fall Chinook along the outer coast, as fall Chinook typically migrate closer to the coast than spring-summer Chinook, which spread out much more widely over the Gulf of Alaska. J Pod might encounter Snake River fall Chinook on the outer coast during forays to the west side of Vancouver Island, but would be less likely to encounter Snake River spring-summer Chinook. (NMFS 2018a)

Genetic analysis of the Hanson et al. (2010c) samples indicate that when Southern Resident killer whales are in inland waters from May to September, they consume Chinook stocks that originate from the Fraser River (including Upper Fraser, Mid Fraser, Lower Fraser, N. Thompson, S. Thompson and Lower Thompson), Puget Sound (N. and S. Puget Sound), the Central British Columbia Coast and West and East Vancouver Island. Hanson et al. (2010c) found that the whales are likely consuming Chinook salmon stocks at least roughly proportional to their local abundance, as inferred by Chinook run-timing pattern and the stocks represented in killer whale prey for a specific area of inland waters, the San Juan Islands. Ongoing studies also confirm a shift to chum salmon in fall (Ford et al. 2010a; Hanson et al. 2010a). Although less is known about the diet of Southern Residents off the Pacific coast, the available information indicates that salmon, particularly Chinook, are also important when the whales occur in coastal waters.

Krahn et al. (2002) examined the ratios of DDT (and its metabolites) to various PCB compounds in the whales, concluded that the whales feed primarily on salmon throughout the year rather than other fish species. The predominance of Chinook salmon in their diet in inland waters, even when other species are more abundant, combined with information to date about prey in coastal waters (above), makes it reasonable to expect that Chinook salmon is equally predominant in the whales' diet when available in coastal waters. It is also reasonable to expect that the diet of Southern Residents is predominantly larger Chinook when available in coastal waters. The diet of Southern Residents in coastal waters is a subject of ongoing research.

4. ENVIRONMENTAL BASELINE CONDITIONS

The environmental baseline includes the past and present impacts of all federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). These factors affect the species' environment or critical habitat in the action area. For the environmental baseline, USEPA conducted two different analyses.

4.1. Status of the Environment- Water Quality

The water quality of the Snake River below the Hells Canyon Dam complex has been altered by anthropogenic forces. The emplacement of the dam complex as well as widespread agriculture up above the complex, contribute to excess nutrient enrichment, inputs of pesticides and other toxics, sediment mobilization downstream, and alteration of water temperature.

Temperature alterations affect aquatic biota metabolism, growth rate, and disease resistance, as well as the timing of adult salmonid migrations, fry emergence, and smoltification. Summer temperatures above 16°C put fish at greater risk of effects that range from effects on the individual organism to effects on the aquatic community level. These effects impair salmon productivity from the reach scale to the stream network scale by reducing the area of usable habitat and adversely affecting fish growth, behavior, and disease resistance. The loss of vegetative shading is the predominant cause of elevated summer water temperatures in tributary networks. Smaller streams with naturally lower temperatures that are critical to maintaining downstream water temperatures are most vulnerable to this effect. The same factors that elevate summer water temperature can decrease winter water temperatures and put salmon at additional risk. Widespread channel widening and reduced base flows further exacerbate seasonal water temperature extremes.

Pollutants also degrade water quality. ESA-listed fish species require clean gravel for successful spawning, egg incubation, and emergence of fry. The effects of pesticides and fertilizers, especially nitrates, on water supplies and aquatic habitats are a significant concern. Water pollution of almost every category is increasing, as are hazardous waste emissions, air pollution, toxic releases, and waste generation (Risser, 2000).

4.1.1. <u>Snake River Hells Canyon TMDLs and 303(d) Lists of Impaired Waters</u>

Section 303(d) of the federal Clean Water Act and 40 CFR §130.7 require states to identify water bodies that do not meet water quality standards and are not supporting their designated uses. These waters are placed on the Section 303(d) List of Water Quality Limited Segments (also known as the list of impaired waters). As required by the CWA, IDEQ periodically conducts a comprehensive analysis of Idaho's water bodies to determine whether they meet state water quality standards and support beneficial uses. IDEQ prepares an "Integrated Report" to list the current conditions of all state waters [CWA 305(b)] as well as those waters that are water quality limited or impaired [CWA 303(d)]. IDEQ lists streams or lakes as impaired for either failing to meet their designated beneficial uses, or for exceeding state water quality criteria. Individual stream reaches are listed for parameters such as water temperature, sedimentation/siltation, fecal coliform, ammonia, oil and grease, dissolved oxygen, toxics, etc.

Waters listed under the CWA 303(d) for temperature are widespread and include the Snake River mainstem below the Hells Canyon Dam. Reach-specific 303(d) listed stream segments are available at: http://www.IDEQ.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report/. The Lower Snake River segment is currently (IDEQ approve 2014 list) listed as impaired for chlorophyll a, DO, TDG, temperature, and methyl mercury (IDEQ web viewer accessed December 10, 2018, https://mapcase.deq.idaho.gov/wq2014/). On the State of Oregon's (Oregon Department of Environmental Quality or ODEQ) ODEQ's 2012 integrated report, this section of the Snake River is listed in category 4a, TMDL approved, TMDL not needed (https://www.oregon.gov/deq/FilterDocs/2012sumAssessment.pdf).

In 2004, USEPA approved 4 TMDLs for the Snake River Hells Canyon Subbasin (17060101, 17050103, 17050115, and 17050201), which includes the action area. The TMDLs addressed bacteria, nutrients, nuisance algae, dissolved oxygen (DO), pesticides, pH, sediment, temperature, and total dissolved gas (TDG) to protect and restore cold water aquatic life, primary contact recreation, domestic water supply, and salmonid spawning designated uses, although it recommended delisting for bacteria and pH. In 2010, additional TMDLs for this subbasin (17060209 and 17060101) addressing temperature, sediment, and bacteria issues in certain tributaries to the Lower Snake River were also approved by USEPA.

4.1.2. Permitted Discharges

Idaho: None

Oregon: Hells Canyon Dam, ODEQ NPDES Permitted discharge USEPA ID OR0027278. Permit expired on 7/31/2009 and has been administratively extended. Discharge is non-contact cooling water and sump leakage.

4.1.3. Current Temperature Conditions

4.1.3.1. Temperatures of the Hells Canyon action area

The following graphs from the IPC (2018) illustrate the current annual temperature regime at Hells Canyon Dam using data monitoring for inflow at Brownlee Reservior (1996-2017) and outflow at Hells Canyon Dam (1991-2017) (Figure 4.1and Figure 4.2). Summer water temperatures are elevated in the action area throughout the summer months into the fall. Water depth and summer stratification in Brownlee Reservoir results in contributions of colder water to the downstream reach through the summer.

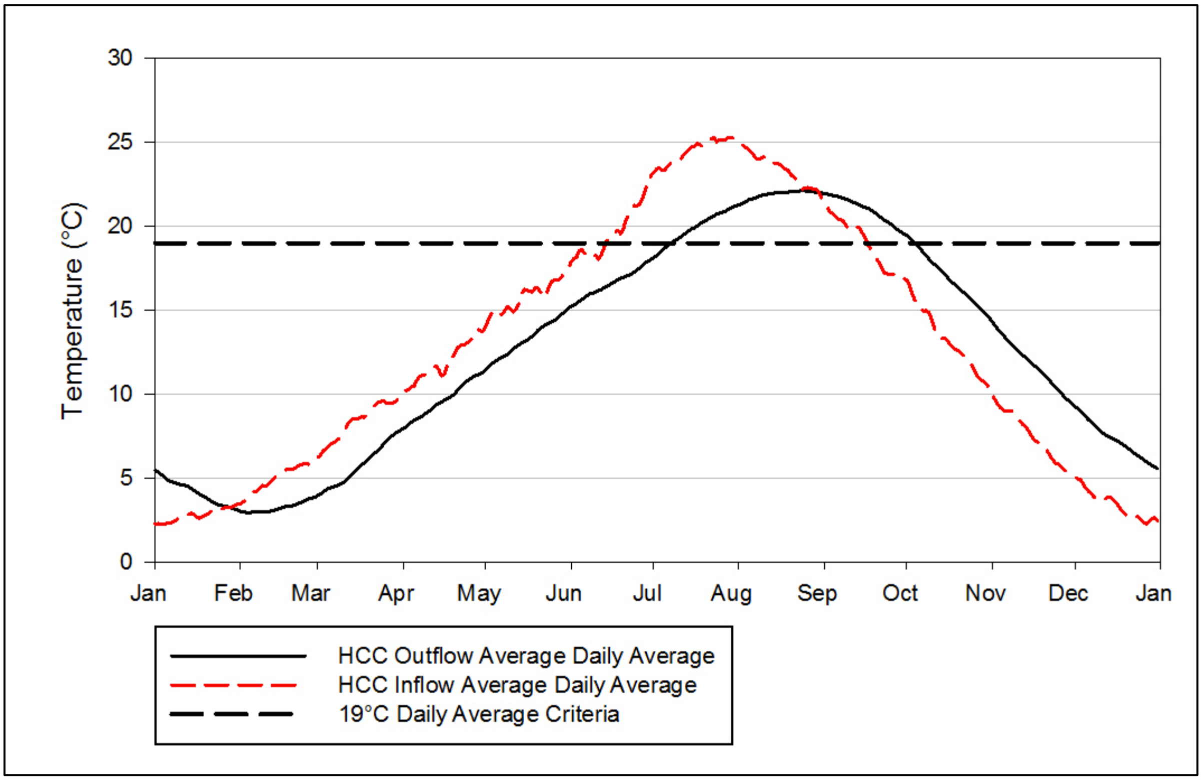

Figure 4.1. Average daily temperatures for the inflow from Brownlee Reservoir (1996-2017) and the outflow at Hells Canyon Dam (1991-2017) (Source: IPC 2018 Figure 6.5-1). Horizontal Line is the Idaho daily average of 19°C.

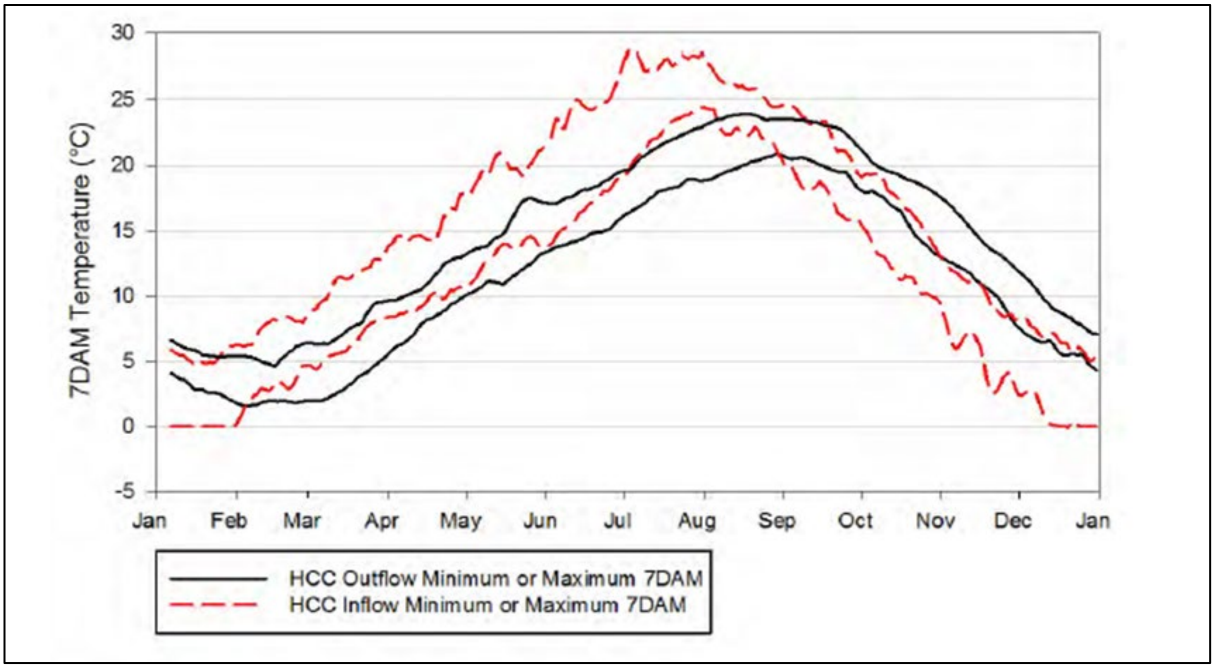

Figure 4.2. Minimum and maximum 7DAM temperature for Brownlee Reservoir inflow (minimum or maximum, 1996-2017) and Hells Canyon Dam outflow (minimum or maximum, 1991-2017) (Source: IPC 2018, Figure 6.1-7).

4.1.3.2. Temperatures experienced by migrating Snake River salmon

Snake River salmon species are exposed to a range of temperatures on their long migration from ocean to freshwater spawning areas. A study by Keefer et al. (2018) looked at thermal regimes of salmon from freshwater entry to the Lower Granite Dam. The following figure is taken from Keefer et al. 2018 shows the range of fish body temperature over the course of the 470 rkm migration from Bonneville to the LGD for individual fish (Figure 4.3 from Figure 8 in Keefer et al. 2018).

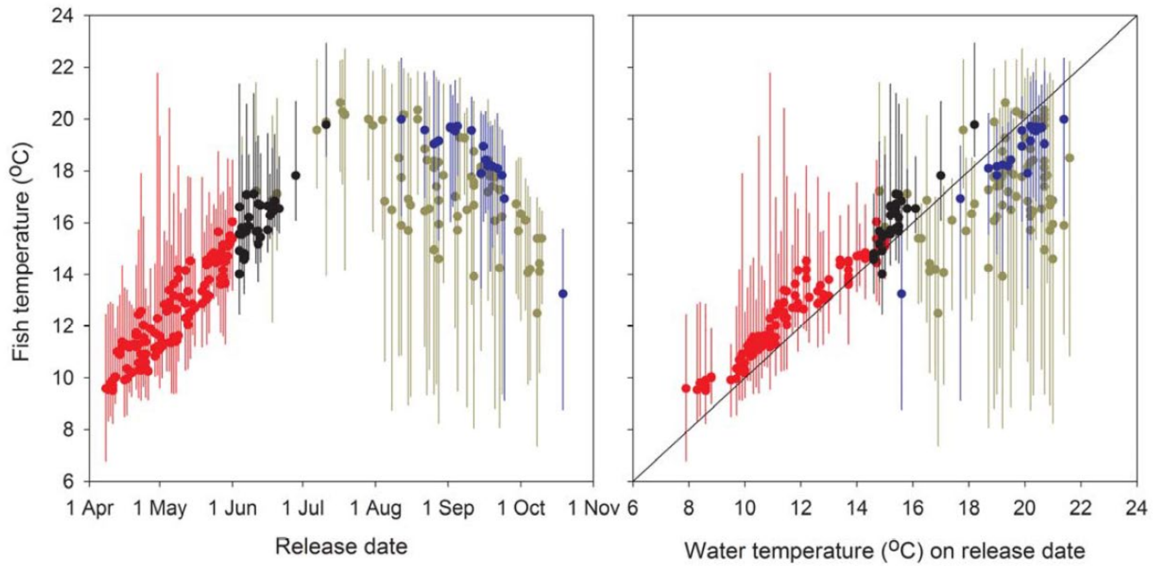

Fig 8. Range of fish temperature exposure over their full migration. Individual Snake River Chinook salmon and steelhead mean, minimum, and maximum body temperatures during the ~470 rkm migration from release to Lower Granite Dam in relation to release date (left panel) and Columbia River water temperature on their release date (right panel) in 2000 and 2002. Red symbols are for spring Chinook salmon (*n* = 126); black symbols are for summer Chinook salmon (*n* = 34); blue symbols are for fall Chinook salmon (*n* = 21); and gold symbols are for summer steelhead (*n* = 75).

Figure 4.3. Snake River salmon/steelhead temperature exposure over range of migration from being tagged and released at Bonneville through to reaching LGD (2000 and 2002). Temperature exposure is shown in relation to release date (panel 1) and to water temperature at release (Source: Figure 8 in Keefer et al. 2018).

USEPA generated a longitudinal temperature profile of the Snake River downstream of the Hells Canyon Dam and Snake tributary data, during the fall spawning period, to analyze temperatures for spawners. Data used for the mainstem Snake River below the Hells Canyon Dam was from Idaho Power monitoring (Idaho Power Company unpublished data, received February 2018). Using temperature data provided by Idaho Power (Unpublished data), maximum weekly (7-day average) maximum temperatures (MWMT) were calculated. Two data checks were implemented on this water temperature data: 1) The data was checked for “missing” data, with only MWMT estimates calculated with at least 4 days of data included in the analysis; 2) In addition, sampling data was removed from the analysis if more than 4 days of data were missing during the first week of the assessment period. The second check was done because water temperatures was almost always warmest during the initial part of the assessment period. Results of this analysis are in Appendix A (Tables 1 and 3).

Data used for the tributaries was from an evaluation of temperature data from the U.S. Geological Survey National Water Information System (NWIS; http://waterdata.usgs.gov/nwis) conducted by Isaak et al (2012). Their analysis included only sites that had multiple years of stream temperature measurements and only those years of site data were retained that had at least 300 daily observations. The time-series of remaining years with data were then examined and only those sites having at least 20 of the 30 years in the period from 1980–2009 were retained (average number of years was 26). Imnaha River, Salmon River and Clearwater River water temperatures were obtained from USFS NorWeST database (www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html#). Sampling locations used in this tributary analysis were the most downstream sampling location with water temperature data during the analysis period. Similar data checks were done to these datasets. The estimated Snake River RM (River Mile) associated with these tributary confluences was estimated from the NHDPlusV1 stream database (www.horizon-systems.com/NHDPlus/NHDPlusV1_home.php). Results of this analysis are in Appendix A (Tables 2 and 4).

Observed water temperatures downstream of the Hells Canyon Dam are well above the current fall spawning water quality standard (13°C maximum weekly (7-day average) maximum temperature (MWMT)), as well as the proposed new standard (14.5°C MWMT) during the fall spawning period (Figure 4.4). Temperature violations are still shown to occur during later spawning periods (i.e., October 23rd through November 6th) (Figure 4.5). These figures also illustrate that both Imnaha, Salmon and Clearwater River temperatures, at their respective confluences with the Snake River, are much colder than Snake River temperatures. It appears that Clearwater River inputs result in dramatically reduced temperatures in the Snake River during the early fall period (i.e., Figure 4.4), while it appears that the Salmon River inflow may be more impactful during the late fall period (i.e., Figure 4.5), and the Imnaha River discharges do not appear to be influential on mainstem Snake River Temperatures during both the summer and fall periods.

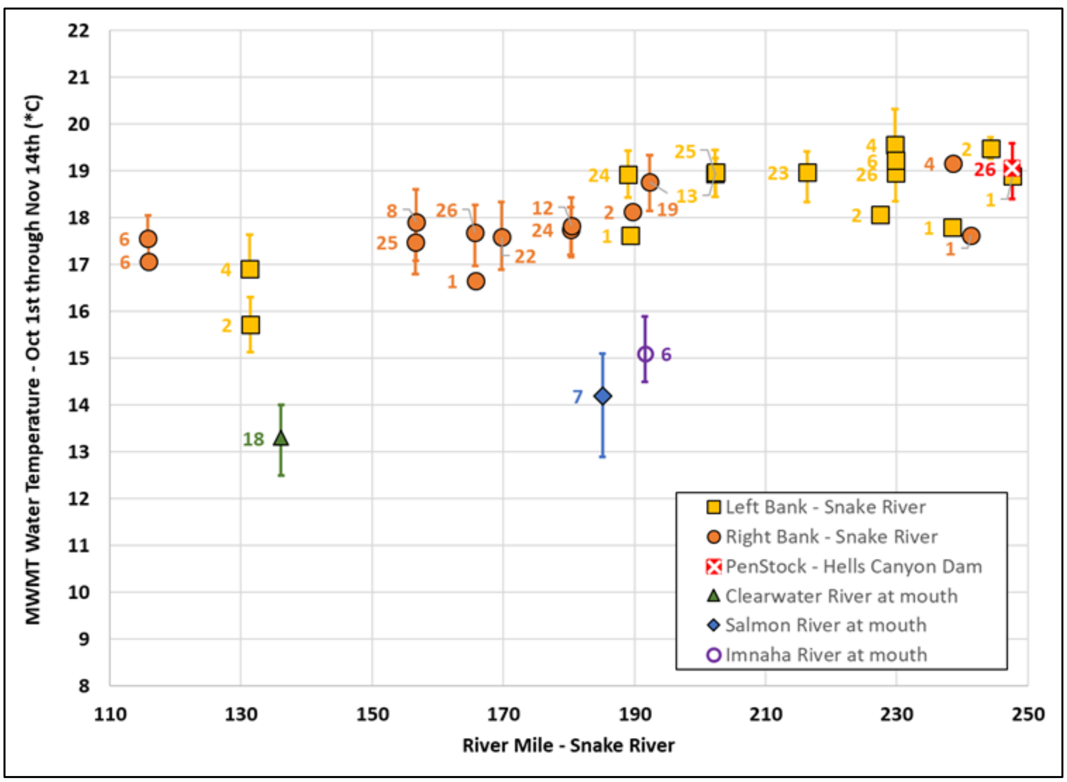

Figure 4.4. Longitudinal Snake River Temperature Profile downstream from the Hells Canyon Dam to the Washington State line observed between 1992 and 2018 for <u>the 10/1 through 11/14 period</u>. Bars represent the 75th and 25th percentile values, and numbers represent the years of data (Source: Idaho Power Unpublished data for mainstem and NorWest data for tributaries to the Snake River).

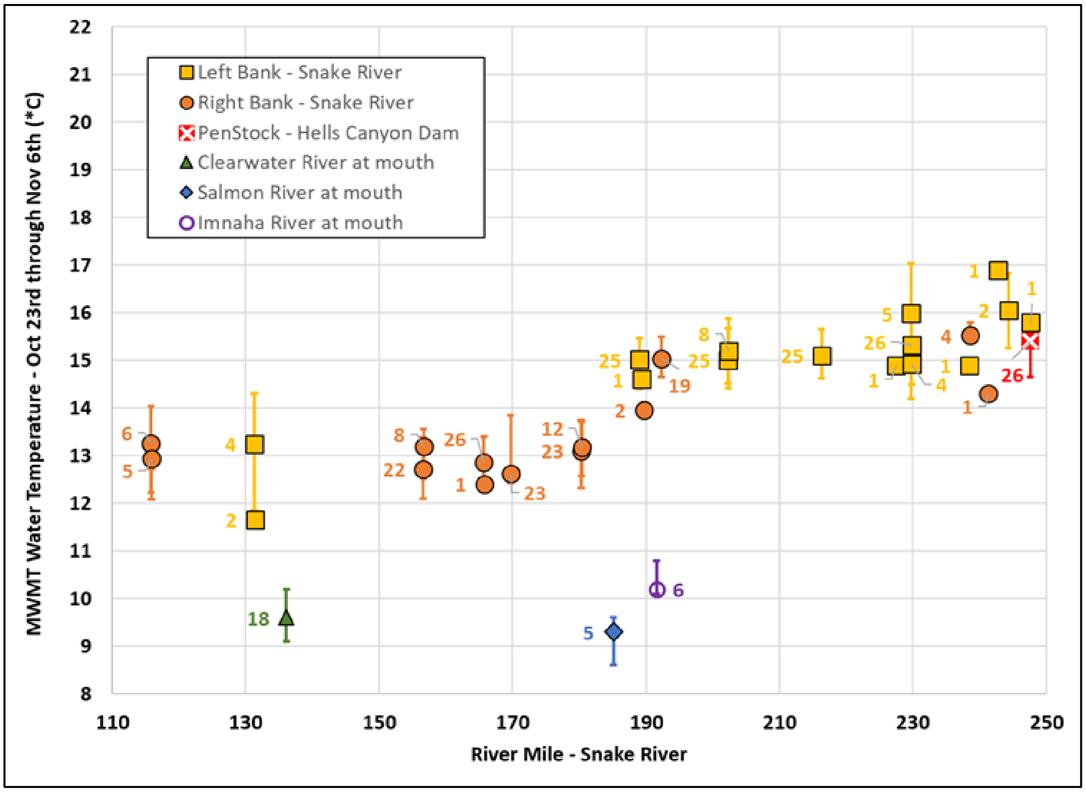

Figure 4.5 Longitudinal Snake River Temperature Profile downstream from the Hells Canyon Dam to the Washington State line observed between 1992 and 2018 for <u>the 10/23 through 11/6</u> period. Bars represent the 75th and 25th percentile values, and numbers represent the years of data (Source: Idaho Power Unpublished data for mainstem and NorWest data for tributaries to the Snake River).

Temperature data for the Imnaha River were also reviewed, as this is important bull trout habitat in the action area. Data were available for 2012, 2014, and part of 2013 (Unpublished data Idaho Power). Coupled with PIT tag detection data from adults migrating out of the Imnaha in the fall, the temperatures of this migration activity can be determined. Imnaha River Maximum Weekly (7-Day Average) Maximum Water Temperatures and bull trout downstream migration counts out of the Imnaha River during the fall are show in (Figure 4.6).

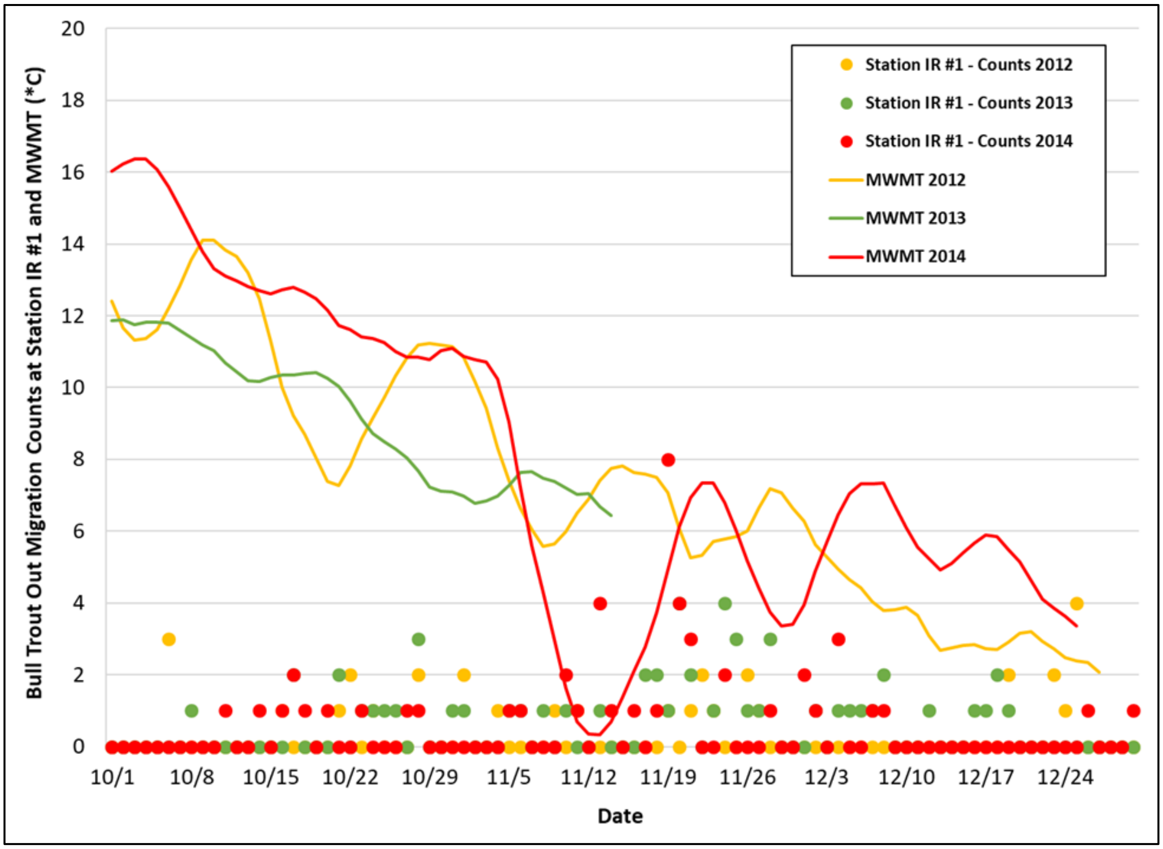

Figure 4.6. The total number of new daily Bull Trout downstream migration counts out of the Imnaha River and calculated Maximum Weekly (7-Day Average) Maximum Water Temperature (MWMT) statistics in the Imnaha River for the fall period of 2012 through 2014 (Source: Idaho Power, Unpublished Data, transmitted by Rick Wilkinson, Idaho Power Company in 2019, Summarized in Appendix C).

4.1.4. Dissolved Oxygen Conditions

The information provided in this section was directly obtained from USEPA 2017c. Dissolved oxygen is essential for the respiration of most aquatic organisms. Reduced oxygen levels have been shown to cause lethal and sublethal effects (physiological and behavioral) in a variety of organisms, especially in fish. Physiological studies indicate that reduced DO levels restrict the ability of fish to maximize metabolic processes (Birtwell 1989). Consequently, the growth rates of fish are affected by reduced DO levels; reductions in the growth rate of salmon have been recorded at levels as high as seven mg/L (EPA 1986a). Sockeye salmon showed signs of elevated blood and buccal pressure and an increased breathing rate at concentrations below 5.07 mg/L (Randall and Smith 1967).

As oxygen availability is reduced in the aquatic environment, fish respond by attempting to maintain oxygen uptake by modifying their behavior, including avoidance, reduced feeding, and reduced swimming capacity. Under simulated estuarine conditions, juvenile Chinook salmon avoided DO levels <7 mg/L (Birtwell 1989). For the coho salmon, DO concentrations lower than 4.5 mg/L caused erratic avoidance behavior (Whitemore et al. 1960). Reduced maximum swimming speeds were observed in coho and sockeye salmon below the ranges of 11.3% (9.17 mg/L) and 9.17% (8.53 mg/L), respectively (Davis et al. 1963; Brett 1964). Reduced disease resistance and fecundity have also been reported for fish living under depressed DO conditions (Davis 1975, Sprague 1985).

Solubility of oxygen decreases as temperature increases and decreases with decreasing atmospheric pressure associated with elevation or barometric change of weather. High water temperature, which reduces oxygen solubility, can compound the stress on fish caused by marginal dissolved oxygen (Bjornn and Reiser 1991). As with other constituents (e.g., metals, suspended solids, and temperature) the early life stages of fish (egg, embryo, alevin) are the most sensitive life stage to alterations of dissolved oxygen. Juvenile salmonids may be able to survive when dissolved oxygen concentrations are relatively low (< 5 mg/L), but growth, food conversion efficiency, and swimming performance will be adversely affected.

While many adult stages of fish can survive at relatively low DO concentrations, the survival of embryos and larvae often requires much higher levels (Welch 1980). For most aquatic species, the time to hatching increases, growth decreases, and survival decreases as DO decreases, with the greatest reduction in survival observed at approximately 5.0 mg/L (Carlson and Siefert 1974; Carlson and Herman 1974; Siefert and Spoor, 1973). In addition, reductions in DO decrease swimming performance in both adult and larval fish (Davis et al., 1963) affecting a species' ability to migrate, forage and avoid predators.

The early life stages of fish are recognized as being the most sensitive and requiring relatively high DO concentrations. The oxygen demand by embryos depends on temperature and on the stage of development, with the greatest DO required just prior to hatching. At near 15°C, intergravel dissolved oxygen (IGDO) requirements for steelhead can exceed 10 mg/L (Rombough 1986, Carlson et al. 1980). Rombough (1988) and other researchers have shown that critical oxygen concentration increases with temperature and with the stage of development of the fish. At 15° C, the critical level of DO (where ambient levels meet metabolic needs) for steelhead increases from 1.0 mg/L shortly after fertilization to greater than 9.7 mg/L prior to hatching.

The crucial timing of IGDO, stream temperature and flow rate varies with each salmonid ESU's specific characteristics. Sowden and Power (1985) observed that survival in field studies is negligible when IGDO falls below 5 mg/L. Phillips and Campbell (1962) observed no survival in a field study where IGDO fell below 8.0 mg/L. They suggest that embryos of newly-produced fry at moderately reduced oxygen levels may not survive well in nature.

DO water quality criteria have been established to protect communities and populations of fish and aquatic life against mortalities as well as prevent adverse effects on eggs, larvae, and population growth. The State of Idaho lists a cold-water biota use standard for DO of exceeding 6 mg/L at all times, and there is a 6.5 mg/L SSC for the Hells Canyon Reach of the Snake River. Idaho’s DO criteria for the salmonid spawning use are presented as absolute minimums, statistical criteria, and as percent saturation. Statistical criteria take into account both short- and long-term exposure to reduced oxygen. DO criteria for salmon spawning apply to specific areas and times of the year. Idaho's salmonid spawning IGDO criteria are: at least 5 mg/L as a 1-day minimum; and at least 6 mg/L as a 7-day mean. The salmonid spawning water column criterion is the greater of: at least 6 mg/L as a 1-day minimum, or 90 % saturation. The less stringent DO criterion (> 6 mg/L at all times) for cold water biota applies before spawning and after developing young fish leave the redd.

EPA (1986a) recommends assuming a 3 mg/L differential between water column DO and IGDO for criteria development; this implies an IGDO of at least 6.7 mg/L for the Rombough (1988) work. However, Maret et al. (1993) found that surface DO and IGDO were within 1 to 2 mg/L in undisturbed streams; this would imply an IGDO of around 9 mg/L for developing steelhead in the Rombough (1988) study.

The State of Idaho's DO criteria are based on USEPA guidance (the Gold Book, EPA 1986b). During the time that salmonid-designated waters support embryo and larval stages, USEPA recommends a water column DO of 11mg/L for no production impairment, 9 mg/L for slight production impairment, and 8 mg/L for moderate production impairment. Assuming the 3 mg/L surface to gravel differential (as described above), the IGDO levels are 8 mg/L, 6 mg/L and 5 mg/L, respectively. Idaho has the 7-day mean IGDO criterion of 6 mg/L, which is well below the 8 mg/L "negligible survival" IGDO level discussed above. There are no 5- or 7-day average criteria for salmonid spawning water column DO or for cold water biota protection in Idaho.

Data from the Hells Canyon Complex outflow shows regular violations of the magnitude of the Idaho water quality standard for DO. Daily data averaged across years 2004-2017 shows mean DO exceeded consistently the 6 mg/L cold-water biota standard mid-August through mid-October (Figure 4.7). In some years DO fell below 4mg/L in the fall time-period (Figure 4.8).

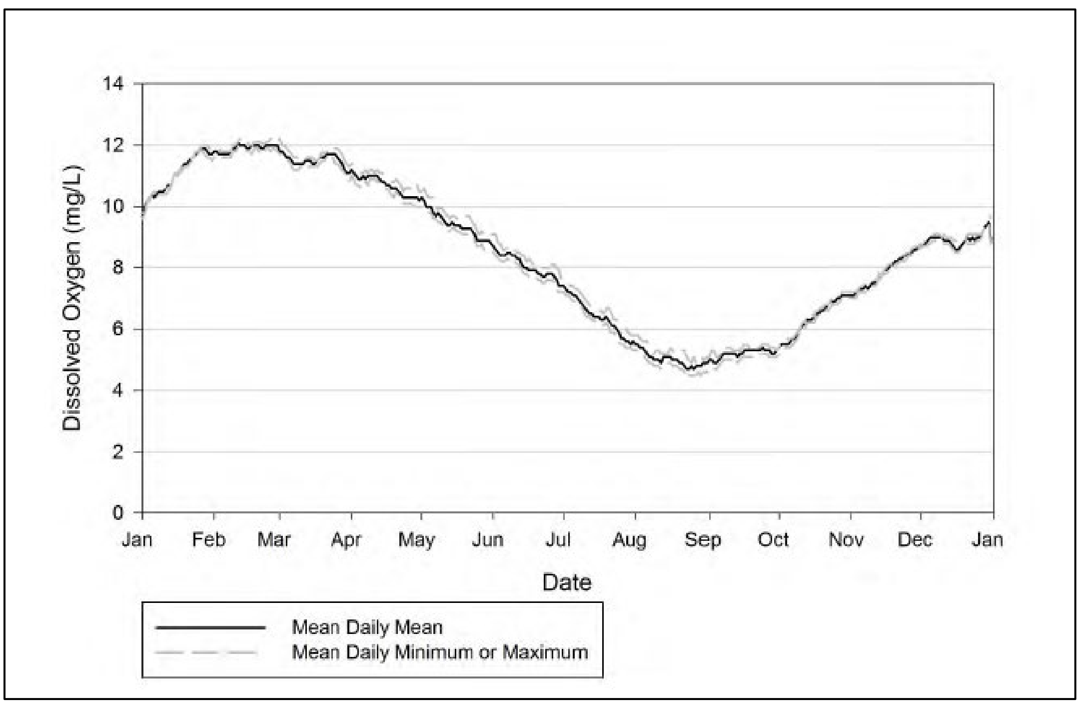

Figure 4.7. Monthy Hells Canyon outflow mean daily mean, mean daily maximum, and mean daily minimum outflow DO summarized from measurements collected at ~10 minute intervals, 2004-2017 (Source Figure 6.2-12 of of IPC 401 Application 2018).

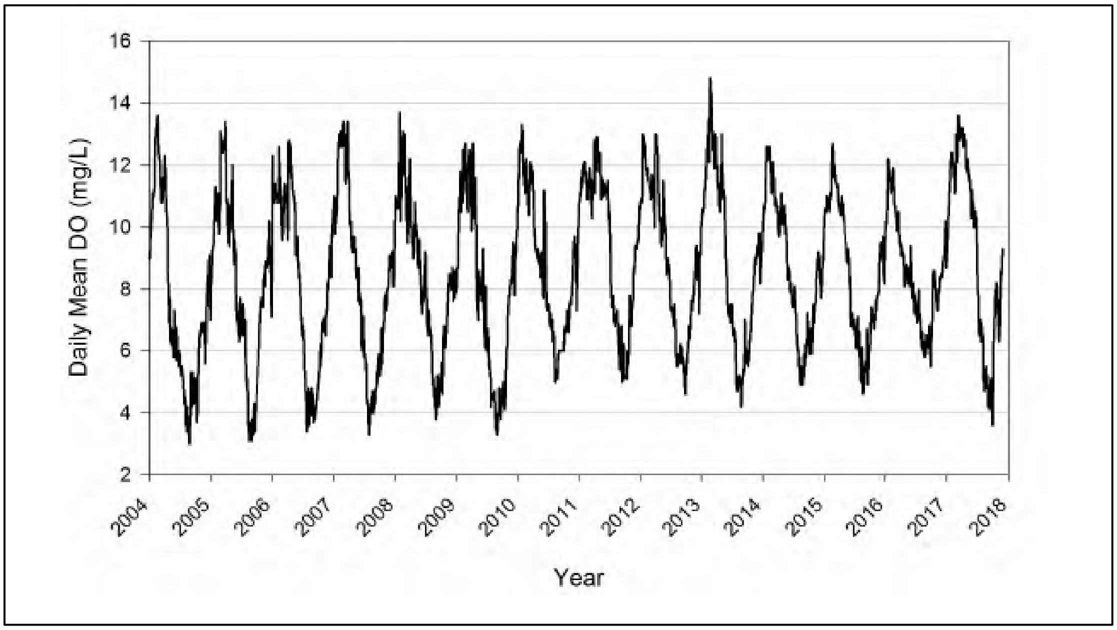

Figure 4.8. Annual Hells Canyon outflow mean daily DO summarized from measurements collected at ~10-minute intervals, 2004-2017 (Source: Figure 6.2-14 of IPC 401 Application 2018).

4.2. General Snake River Hydrosystem Effects to Baseline Conditions

The information provided in this section was directly obtained from the USFWS 2015 Toxics Biological Opinion (USFWS2015). Storage dams have eliminated mainstem spawning and rearing habitat and have altered the natural flow regime of the Snake River, decreasing spring and summer flows, increasing fall and winter flow, and altering natural thermal patterns. Slowed water velocity and increased temperatures in reservoirs delays smolt migration timing and increases predation in the migratory corridor (NMFS 2014; Independent Scientific Group 1996; National Research Council 1996). Formerly complex mainstem habitats have been reduced to predominantly single channels, with reduced floodplains and off-channel habitats eliminated or disconnected from the main channel (Sedell and Froggatt 2000).

As stated in Section 3.4, dams in the basin have had a long-term impact on salmon species. The construction of hydroelectric and water storage dams without adequate provision for adult and juvenile passage in the Upper Snake River has kept fish from all spawning areas upstream of Hells Canyon Dam. The lower reaches of the Columbia River are highly modified by urbanization and dredging for navigation. The upland areas covered by this ESU are extensively logged, affecting water quality in the smaller streams used primarily by summer runs. The construction of the HCC specifically extirpated a large area of spawning habitat (some of the best spawning habitat in the PNW, the Marsing Reach) and, together with other hydrosystem construction in the Columbia Interior Basin, coincided in a dramatic decrease in salmonid populations in the Snake River, Hells Canyon.

In the Snake River, hydrosystem projects create substantial habitat blockages in the Snake River (SR) ESU; the major ones are the Hells Canyon Dam complex (mainstem Snake River) and Dworshak Dam (North Fork Clearwater River). Minor blockages are common throughout the region. In addition to blockage of adult passage upstream, the HCC construction, including Brownlee Dam in 1958, resulted in a thermal shift and resulted in a modified thermal regime that delayed fall cooling, resulting in a delayed spawning season, increased winter base temperatures, delayed the spring warming, and resulted in cooler maximum peak summer temperatures compared to upstream/inflow conditions (IPC 2018). The effects of thermal shifts are directly relevant to the action and are detailed in further sections.

Flows are also altered by the Hells Canyon Complex. The upstream storage impacts the hydrograph, with a relatively constant flow maintained between 8,500 and 13,000 cfs below the Hells Canyon Dam to fully submerge fall Chinook redds during the spawning season (IPC 2018).

4.3. Factors Influencing Conditions for Salmonids

Besides the impacts to ESA-listed species in the Action Area from the development and operation of the hydroelectric facilities, other anthropogenic activities that have degraded aquatic habitats or affected native fish populations in the Snake River basin. These are based on information in the USFWS 2015 Toxics Biological Opinion (USFWS2015). These activities include stream channelization, elimination of wetlands, construction of flood-control dams and levees, construction of roads (many with impassable culverts), timber harvest, splash dams, mining, water withdrawals, unscreened water diversions, agriculture, livestock grazing, urbanization, outdoor recreation, fire exclusion/suppression, artificial fish propagation, fish harvest, and introduction of non-native species (Henjum et al. 1994; Rhodes et al. 1994; National Research Council 1996; Spence et al. 1996; NMFS 2004). In many watersheds, land management and development activities have:

--Reduced connectivity (i.e., the flow of energy, organisms, and materials) between streams, riparian areas, floodplains, and uplands;
--Elevated fine sediment yields, degrading spawning and rearing habitat;
--Reduced large woody material that traps sediment, stabilizes streambanks, and form pools and contributes to hydraulic diversity.
--Reduced vegetative canopy that minimizes solar heating of streams;
--Caused streams to become straighter, wider, and shallower, thereby reducing rearing habitat and increasing water temperature fluctuations;
--Altered peak flow volume and timing, leading to channel changes and potentially altering fish migration behavior; and,
--Altered floodplain function, water tables and base flows.

Tributaries are important salmonid habitat areas for spawning and rear (except for fall chinook, mainstem spawners). In these systems the integrity of riparian areas is essential to stream function and habitat quality for salmonids. Many of the habitat issues listed above result from the poor management of riparian areas. Healthy riparian areas retain the structure and function of natural landscapes as they were before the intensive land use and land conversion that have occurred over the past 150 to 200 years. However, land use activities have reduced the numbers of large trees, the amount of closed-canopy forests, and the proportion of older forests in riparian areas.

Introduction of nonnative species either as part of sanctioned fisheries management agendas or endeavors by private individuals has been ongoing in the west for many decades. The objectives of planting fish vary (e.g. enhance fishing experiences, influence food web) and the outcomes to a particular waterbody can also vary. In the west, existence of non-native aquatic species is common in lotic habitats. Using a random sampling design to provide west-wide estimates of the condition of rivers and streams, Stoddard et al. (2005) estimated non-native fish and/or amphibians were common (i.e., they represented more than 10 % of individuals collected at a site) in approximately 34% of the stream and river length in the Western U.S. Restricting this dataset to the interior Columbia Basin, Herger et al. (2007) estimated 26% of the wadeable streams held non-native fish species. Of these, brook trout were the most common non-native species. The results of an introductions ecological community and the integrity of the existing fish assemblage can range from no measurable effect to the ecological function of a system or to situations that can be catastrophic for the fish assemblage. For example, negative impacts of planted non-native brook trout to native bull trout are well known.

4.3.1. Cold Water Refuges

Cold water refuges are patches of cooler water that are available to coldwater species. This habitat feature is important for salmon and trout to allow for instream movements and migration during periods of elevated temperatures without significant affects. These patches of relatively colder water were defined by Ebersole et al. (2015) as ≥3°C colder than the ambient stream temperature. The occurrence of cold water refuges in mainstem rivers such as the Snake River are spatially complex and are influenced by the contributed of cold water from a variety of sources; confluences with colder tributaries, inputs from small perennial streams, input from groundwater upwelling, and subsurface flows from intermittent and ephemeral channels (Ebersole et al. 2015). As discussed by Fullerton et al. (2015), determining if adequate cold-water refuges are available is a complex question because there are many factors to consider including: the size of the cold-water patches, the distribution and frequency of the patches, and whether they are available at the actual time-periods and actual locations when they are needed by the fish.

There has been some work to evaluate the occurrence of cold water refuges in the Hells Canyon reach. The IDEQ/ODEQ 2004 TMDL found that bull trout and steelhead, which can be present in the summer and fall months in this section of the Snake River escape through the multiple colder tributaries available as refuges (IDEQ 2004). A study by Chandler et. al (2003) showed that the rainbow trout populations in the HCC and rainbow trout and bull trout downstream were using cold-water refugia provided by the tributaries during summer months by either migrating upstream into the tributaries or associating with the cold-water plume of the tributaries during the summer months. Using the State of Oregon definition of Cold-water Refugia (CWR) of ≥2°C colder than the ambient (i.e. mixed) water, Idaho Power (2018, Exhibit 6.1-1)) evaluated temperature data from Hells Canyon corridor streams to identify the likely contributions of cold water. They compared temperature data from two sets of perennial streams (low and high elevation headwaters) and the Imnaha River to Snake River temperature data collected at two locations.

Results from the low elevation headwater data set compared to the Snake River demonstrated that CWR was provided during at least some portion of the day, except for a few days in the middle of July, where all diel metrics exceeded the -2°C CWR definition (Idaho Power 2018, Section 6). Generally, by mid-August, the daily average temperatures started to drop below the -2°C CWR definition. They concluded that this suggests that thermal refugia was available over the majority of the diel cycle. Finally, by around the first of September, all of the tributaries provided CWR during the entire diel cycle. Results from the higher elevation headwater tributaries demonstrate a much colder pattern relative to the Snake River. All of the tributaries with the exception of Temperance Creek and Kirkwood Creek provide significant CWR during all portions of the diel cycle. Besides these perennial streams the SR-HC TMDL identified 813 drainages classified as intermittent. The contribution of these to providing cold water patches has not been evaluated but may be another source of thermal refugia for salmonids in this reach (Idaho Power 2018).

4.3.2. Issues Related to Hatchery inputs of Chinook

The 2017 fall Chinook recovery plan (NOAA 2017) provides abundant information on the issue of hatchery fish impacts to the ESU. The threats are summarized as two primary issues: 1) the high proportion of hatchery fish as juveniles resulting limiting factors of competition with wild fish for habitat, food, and other resources, 2) high proportion of hatchery-origin spawners resulting in limiting factors of genetic change, loss of fitness, competition among spawners for resources, including spawning areas. The following section from the Recovery Plan (NOAA 2017 is Section 5.5.1, which describes the abundance of hatchery fish as part of the population of the Snake River fall Chinook:

"As described in Chapter 4, based on estimates made at Lower Granite Dam, the proportion of natural-origin fish in the population from 2007 to 2016 has averaged only 30 percent, based on post-harvest, post-broodstock collection estimates above Lower Granite Dam (Young, personal communication, 2017). However, during the same period, annual abundance of natural-origin fish was in the thousands, which represents a dramatic improvement over abundance levels in the 1990s.

There are several possible contributing causes to the increased abundance of Snake River fall Chinook salmon, including reduced harvest rates, improved in-river rearing and migration conditions, and the development of life-history adaptations to current conditions. In some year, improved ocean conditions are thought to be beneficial (Cooney and Ford 2007). However, these conditions fluctuate over the years and in some years may be detrimental to fall chinook abundance. Snake River fall Chinook salmon hatchery programs have also grown through time. Undoubtedly, there are more natural-origin fish present now than before the hatchery programs began, but it is not possible to determine how much of this increase in natural-origin abundance is due to a real growth in natural productivity rather than a consequence of more natural-origin fish being produced simply because the hatchery programs have artificially put more fish on the spawning grounds. In the 1990s, the hatchery programs were limited by measures imposed to reduce the inclusion of stray fish from other programs. With these limitations no longer in place, the hatchery programs have now reached their full intended sizes, and under this more stable situation, the relative contribution of the hatchery programs to abundance and other factors should be easier to determine (NMFS 2012a).

4.4. Alterations to Fall Chinook Habitat Following Dam Emplacement

HCD was completed in 1968 and became the upstream terminus for salmon migration. Spawning habitats in the lower Snake River were lost with the construction of the federal Lower Snake River dams, beginning in 1962 with the completion of Ice Harbor Dam and going through 1975 with the completion of Lower Granite Dam. This construction further limited spawning in the Snake River to the approximately 100 miles of free-flowing river between HCD and Lower Granite Reservoir. Snake River Fall Chinook (SRFC) life history is intrinsically linked to the thermal regimes in the Snake River and these thermal regimes have been significantly altered from the historical condition by the construction of the Hells Canyon Complex. The following sections detail temperature related habitat alterations and the resulting conditions relevant to this BE.

4.4.1. Overview

SRFC salmon have a varied history of different thermal regimes. Adults migrate in late summer and early fall when summer maximum temperatures are at or near their peak. They spawn during a declining thermal pattern in the fall. These thermal regimes vary among years and spawning locations, influenced by differences in water year and air temperatures. After the installation of the Hells Canyon complex of dams the thermal regime of the Hells Canyon shifted resulting in in warmer fall and winter temperatures relative to the pre-HCC thermal regime. Although this reach was primarily a migration corridor prior for SRFC to the HCC, the shift in thermal regime has resulted in habitat that is conducive to spawning and incubation for SRFC. The thermal environment below the HCC now supports incubation and emergence timing similar to those historically upstream of the HCC, whereas historically the HCC was a colder incubation environment that would have delayed emergence timing. Adequate spawning gravels are available below HCD and the reach has relatively low quantity of fine sediment due to capture of fines in the HCC reservoirs.

4.4.2. Historic situation (source: IPC 2018 401 application)

The core population of SRFC salmon historically occupied the mainstem Snake River primarily upstream of Swan Falls Dam. They were closely associated with the warmer winter thermal regime of the Middle Snake River, which was significantly influenced by the discharge of the Eastern Snake Plain Aquifer (ESPA). The thermal pattern of the Snake River is unique from other rivers because of the high volume of groundwater stored in this aquifer. In total, approximately 5,000 cfs of groundwater enters the Snake River (between approximately RM 553 and RM 620) in the form of springs that flow from basalt cliffs. The warmer water inputs from the ESPA translate into warmer incubation temperatures for SRFC resulting in allowance for early emergence from spawning areas followed by a short freshwater rearing phase before outmigration (referred to as Age-0 life history). This life history strategy is dependent on early emergence to allow enough time to feed and grow before then migrating early enough before summer water temperatures become unsuitable. In contrast some Chinook salmon have an Age-1 type strategy, where fish will rear during the first year in freshwater and migrate to the ocean as a 1-year old fish. The thermal regime for Age-1 life histories must be cool enough to support summer rearing, which was not likely in the arid desert environment of the mainstem Snake River.

Prior to the construction of the HCC, the Snake River in Hells Canyon was relatively cold, and fish would have emerged late relative to those upstream in the Swan Falls reach and would have had to rear and migrate during warm summer temperatures. This thermal regime was very similar to the Salmon River, which historically has not supported significant SRFC salmon spawning. When Brownlee Reservoir and Dam were constructed and blocked migration, it also created a thermal shift with warmer fall temperatures. The reservoir also moderated winter temperatures to be warmer than what historically occurred below Brownlee Dam. This new thermal regime created conditions for emergence timing comparable to below Swan Falls Dam and continues today to support the Age 0 life history.

The modification of the thermal regime is illustrated by using the mean daily average temperature of the Snake River measured below HCD for the time-period 1996 to 2006 (Figure 4.9 from Figure 6.1-10 in Idaho Power 2018).Construction of Brownlee Dam (1958) modified the thermal regime in the Hells Canyon reach of the Snake River, causing 1) delayed fall cooling, 2) increased winter base temperatures, 3) delayed spring warming, and 4) cooler summer temperatures relative to inflow conditions.

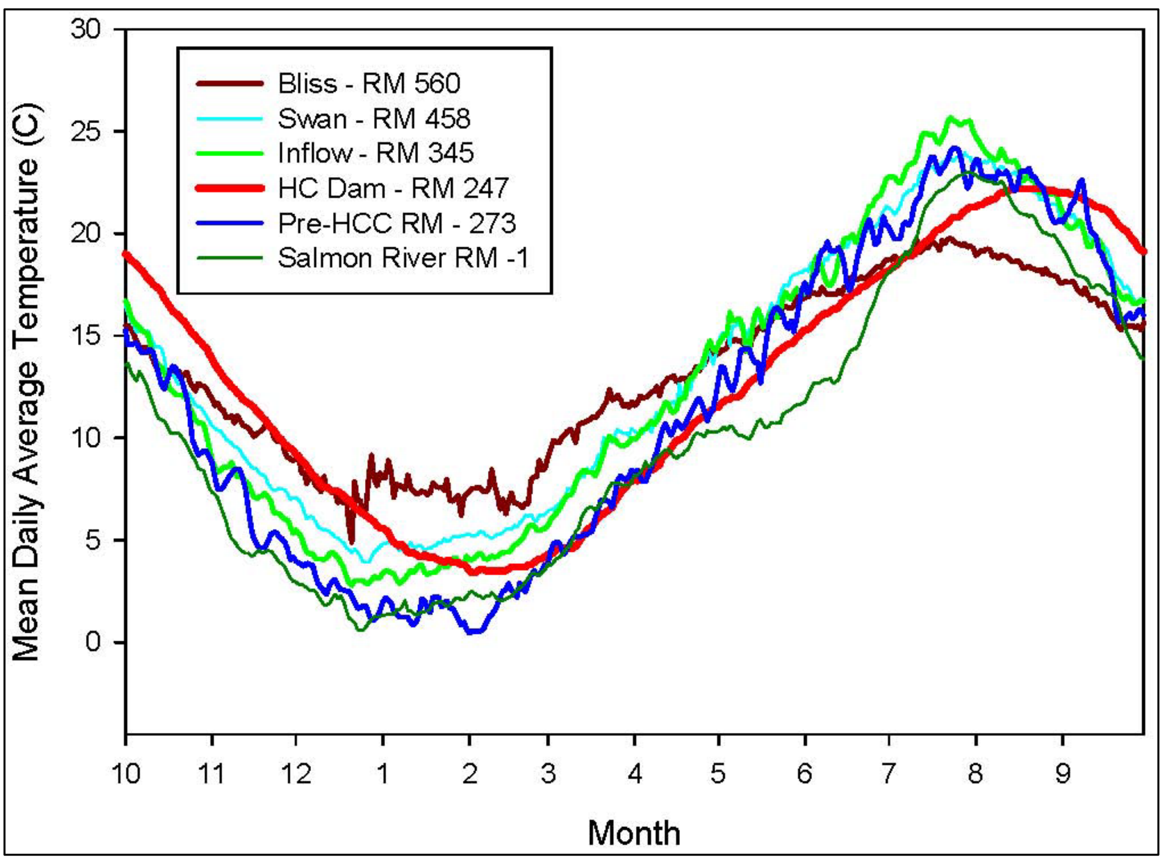

Figure 4.9. Mean daily average water temperature in °C that represents thermal patterns of the Snake River for the time-period 1996–2006 at Bliss Dam (RM 560), Swan Falls Dam (RM 458), a location above the inflow to Brownlee Reservoir (RM 345), HCD (RM 247), and the Salmon River (RM 1) and for the time-period1954–1957 for the pre-HCC location at RM 273 (Source ID Power 2018 Figure 6.1-10).

Water temperatures in the reach typically range from 20°C to 23°C (68°F to 73°F) in early September, fall below 20°C (68°F) in late September, and continue to decline through the month of January (Figure 4.10). The reach does not freeze in the winter, as it sometimes did historically.

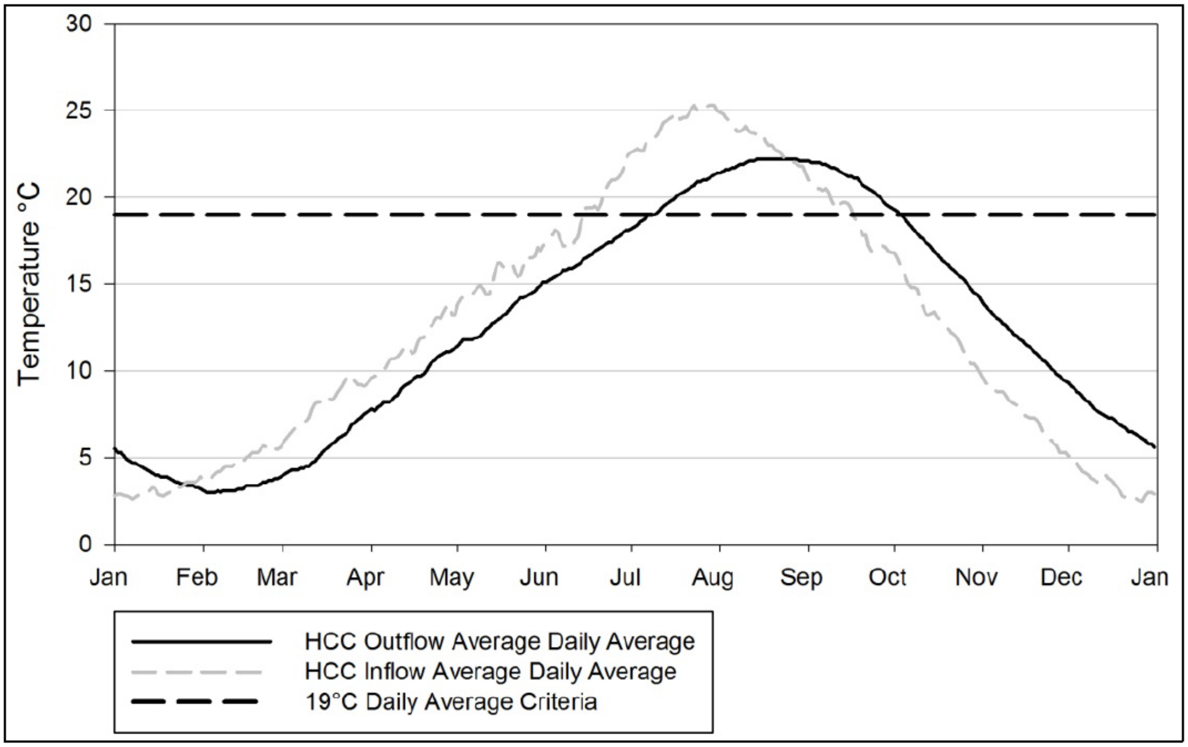

Figure 4.10. Daily average temperature in °C inflow to Brownlee Reservoir and outflow from Hells Canyon Dam for the 1996-2012 period of record compared with Idaho's daily average criteria (Source: NOAA 2017 Fall Chinook Recovery Plan).

4.4.3. <u>Delays to peak summer temperatures (from 2018 recovery plan)</u>

The largest reservoir in the Hells Canyon Complex is Brownlee, at the head of the complex, followed by the Oxbow and Hells Canyon Reservoirs. The Oxbow and Hells Canyon Reservoirs have little storage capacity, so most of the water released from Brownlee travels downstream through the Oxbow and Hells Canyon projects within a day. The general effect is that the large thermal mass created by the water stored in these reservoirs delays the peak summer water temperature to a later date and maintains temperatures at a higher level later into the fall relative to what would occur in a natural river condition.

While the delay in peak temperature is a consistent trend on an annual basis, a subtler effect of reservoir operations on water temperatures exists between years. During wet years, the Hells Canyon Complex of reservoirs is drawn down for flood control. Refill of these reservoirs occurs in the spring when water temperatures have started to warm. Thus, when this water is released in the summer, it creates a warmer river environment below the projects. Conversely, in a dry year the projects are not drawn so deeply during the winter months for flood control, resulting in less refill, and the water in storage is cooler, thus creating a cooler water environment below the projects during the summer when this water is released.

4.4.4. Ramifications of altered thermal regimes--adult spawners (from NOAA 2017 Recovery Plan)

Recent (2007 to 2016) average migration timing of fall Chinook salmon and average daily temperatures at Lower Granite Dam are presented in Figure 4.11. The timing and distribution of adults upstream of Lower Granite Dam is not well known. Fall Chinook salmon thermoregulate by delaying migration and using localized cool water areas (Goniea et al. 2006; Clabough et al. 2018). Some adult fall Chinook salmon ─ especially those migrating past Lower Granite Dam in late August and early September when water temperatures are highest ─ likely hold downstream of the Clearwater River confluence (which is typically cooled below historical temperatures by releases of cold water at Dworshak Dam). The fish probably also hold temporarily downstream of the confluence with the Salmon River, which cools more rapidly than the Snake River (primarily because of Brownlee Reservoir) in the fall, and near other small tributaries.

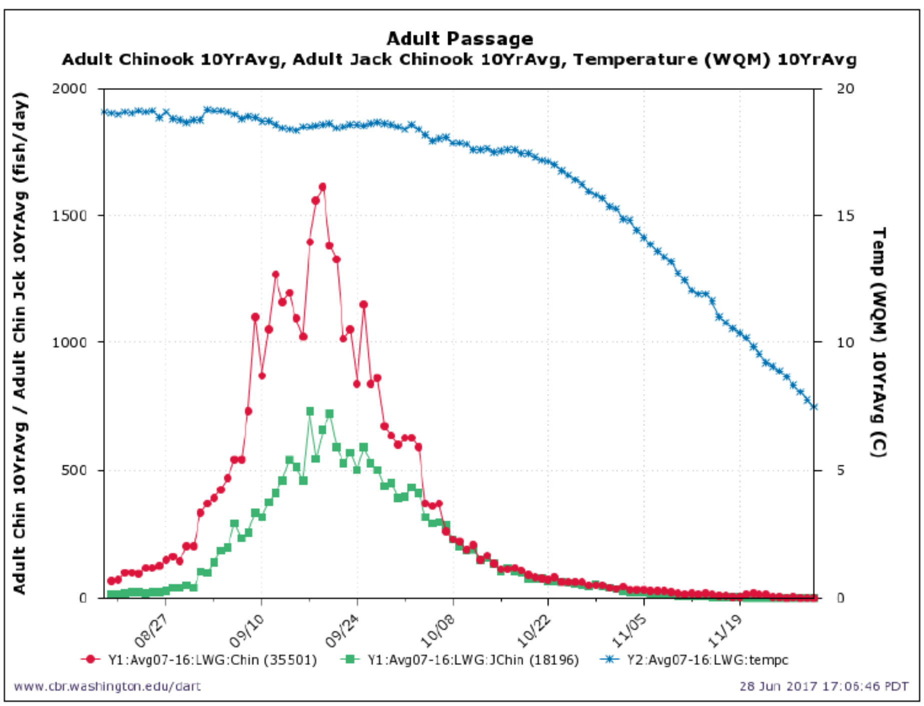

Figure 4.11. Average 10-year migration timing of adult Snake River fall Chinook salmon (red line) and adult jacks (green line) in relation to average 10-year daily temperatures (blue line) at Lower Granite Dam (Source: NOAA 2017 Recovery Plan).

Operation of the reservoirs during the late summer and fall can have a significant effect on the temperatures that adult spawners experience. If inflow to the project is high in the late summer, then the warmer water collected during summer months is discharged and is replaced by cooler water inflow during the fall, which passes downstream, creating cooler river environment. Conversely when inflow to the project is low during the late summer and early fall, the warmer water in the reservoir is maintained, which is passed through the projects, creating warmer water conditions for fall spawning fish.

The effects on adults returning to spawn in late summer/early fall are uncertain but could be negative. About 90 percent of adult fall Chinook salmon pass Lower Granite Dam and enter this reach between late August and early October, but water temperatures in the reach sometimes do not fall below the USEPA-recommend criterion of 20°C (68°F) for migrating adult Chinook salmon until mid-to-late September. Thus, most adult fall Chinook salmon migrating, holding, and spawning downstream of Hells Canyon Dam could be exposed to warmer temperatures for longer periods of time than occurred historically, either in the presently available mainstem Snake River habitat or the habitat formerly accessible upstream. The warmer temperatures could affect fall Chinook salmon abundance and productivity by increasing pre-spawning mortality and reducing spawning success. It is not clear, however, that the temperature regime reduces productivity. It is known that returning adults use cold-water refuges (Keefer et al. 2018), such as near the mouths of the Salmon River and other smaller tributaries.

According to the 2017 NOAA Fall Chinook Recovery Plan, potential impacts on pre-spawning adults from water temperatures in this reach remain uncertain. Available literature on all run types of Chinook salmon (used to develop the 20 °C (68 °F) water quality temperature standard) suggests that adult exposure to the current thermal regime would be associated with some level of either lethal effects (e.g., pre-spawning mortality) or non-lethal effects (e.g., decreased spawning or egg viability) (ODEQ 1995a; McCullough 1999; WDOE 2000a; EPA 2001, 2003; Mann and Peery 2005; Jensen et al. 2005). It is not clear, however, that the current thermal regime is significantly affecting pre-spawning fall Chinook salmon in this reach. Comparisons of adult escapement estimates and fish-to-redd ratios documented in the Snake River do not suggest that substantial numbers of adult fall Chinook salmon are dying prior to spawning as a result of their exposure to elevated fall water temperatures. NOAA (2017) opines the following: It is possible that the size and non-confined nature of the river in this reach below Hells Canyon Dam, a declining thermal regime after August, and opportunities to escape the high temperatures by moving to cool-water refugia (e.g., the confluences of the Clearwater River, Salmon River, and other tributary streams with the Snake River) make the fish less susceptible to disease and mortality than literature and laboratory studies might indicate. Further, the literature is general to all Chinook salmon run types, and there is reason to believe that fall Chinook salmon are more tolerant of higher temperatures than other stocks of Chinook salmon. Nonetheless, in some years, adults passing Lower Granite Dam in late August and early September may still be exposed to 18 to 22 °C (64 to 72 °F) water temperatures for several days or weeks prior to spawning in this reach, and the prolonged exposure of adults to elevated temperatures in the migration corridor and spawning areas could potentially result in reduced spawning success and some egg and fry mortality (Mann and Peery 2005; Jensen et al. 2005, 2006).

In addition, hydrosystem operations, including variable hydraulic gradient and temperature, can also negatively impact the ability of fall Chinook to spawn by altering the cues each species uses to select redd sites, according to a study below Bonneville Dam (Geist et al. 2008). Likewise, there is evidence that the declines in fall Chinook in the Columbia and Snake system over the last 50 years [as of 1999] corresponded to construction and completion of the hydropower system (Schaller et al. 1999).

4.4.5. Ramifications of Altered thermal regimes—incubation and rearing (from NOAA 2017 recovery plan)

The current thermal regime, strongly influenced by Brownlee Reservoir, creates warmer conditions during the egg incubation period. These conditions foster earlier fry emergence and influence the timing of other life-history stages (parr and smolt). The altered thermal regime also favors the historically dominant Snake River fall Chinook salmon subyearling life-history strategy. Compared to historical conditions, the earlier emerging fry feed and grow in shoreline rearing areas and then outmigrate earlier, when water-temperature mediated effects such as increased mortality, disease, exposure to predators, and reduced physiological development are less severe.

4.4.6. Alteration of SRFC spawning and incubation periods from historic (taken from IPC 401 Certification Application 2018).

According to IPC 401 Certification Application (2018): Today, some of the earliest spawning observed in the Snake River is during the second week of October. The peak spawning period (the median distribution of redd observations for the years 1993–2009) is November 4. The latest spawning observations are generally near the second week in December. Evermann (1896) reported observations of ripe and spent fall Chinook salmon in a fishery at Millet Island in 1894. The fishery began on October 1 and extended through October 31. ….Ripe fish were still being captured at the close of the fishery, suggesting spawning continued after November 1. An observation reported by Evermann from an interview with a seine fisherman near Glenns Ferry (RM 539) reported observing carcasses through the first half of November. Similarly, below Swan Falls Dam, Zimmer (1950) reported 3 redds observed in the first week of October 1947, with a peak number of redds counted on the November 6 flight, and spawning was generally completed by the end of the first week in December.

4.5. Comparisons to an Undammed Reach: the Hanford Reach

Upper Columbia River summer/fall Chinook are considered closely related to the Snake River fall Chinook. A portion of the Upper Columbia Chinook ESU that is the highly productive is the Hanford Reach population, which spawns in the Hanford Reach of the Columbia. The Hanford reach of the Columbia River is unique in that it is the remaining free flowing section of the Columbia River. Information provided by NOAA (memorandum from Ritchie Graves, NOAA, to Dan Opalski, USEPA, 2019; Appendix D) on Hanford Reach Chinook run timing is presented in this baseline to demonstrate fall Chinook run characteristics where temperatures are more aligned with a natural (un-dammed condition).

The data sources used by NOAA are in the memo (Appendix D). Data from two reaches are presented; Vernita Bar Reach (Figure 4.12) and for the entire reach (Figure 4.13). The Vernita Bar Reach is the most productive reach within the greater Hanford reach. NOAA concludes that these data show, since at least 2004, this population initiates spawning at temperatures above or near 14.5C.

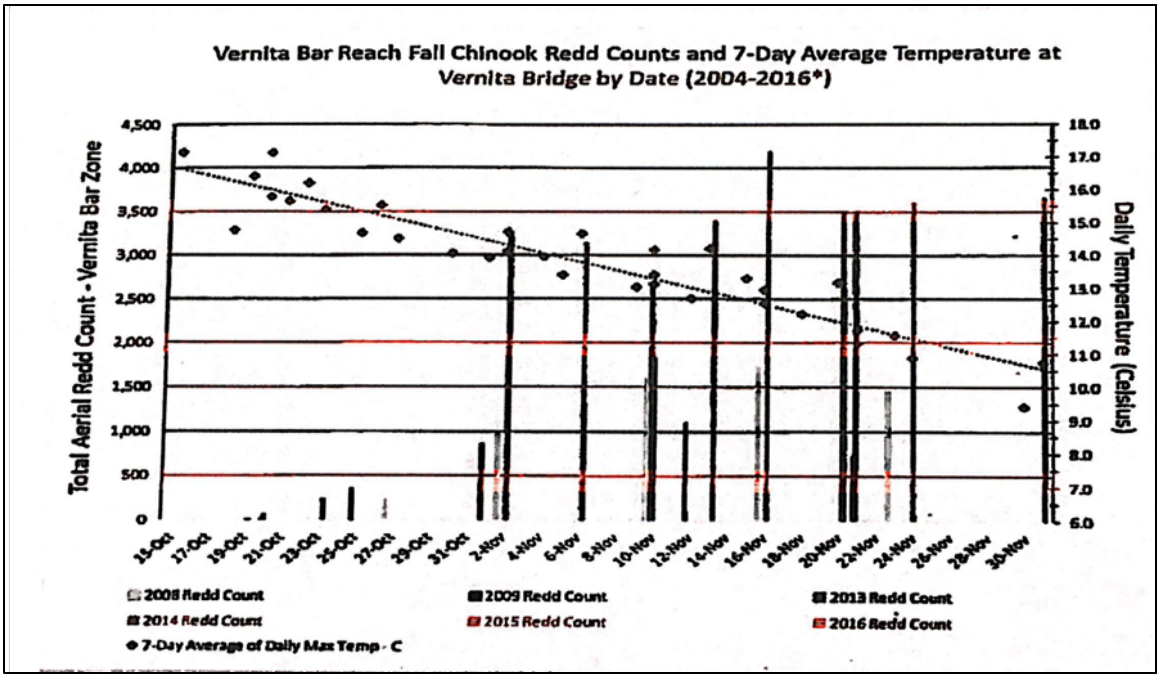

Figure 4.12. Six years of total Chinook redd counts from Vernita Bar Reach between 2004 and 2016. Seven-day average temperature shown as points with regression line (Source: NOAA memo to USEPA; Appendix D).

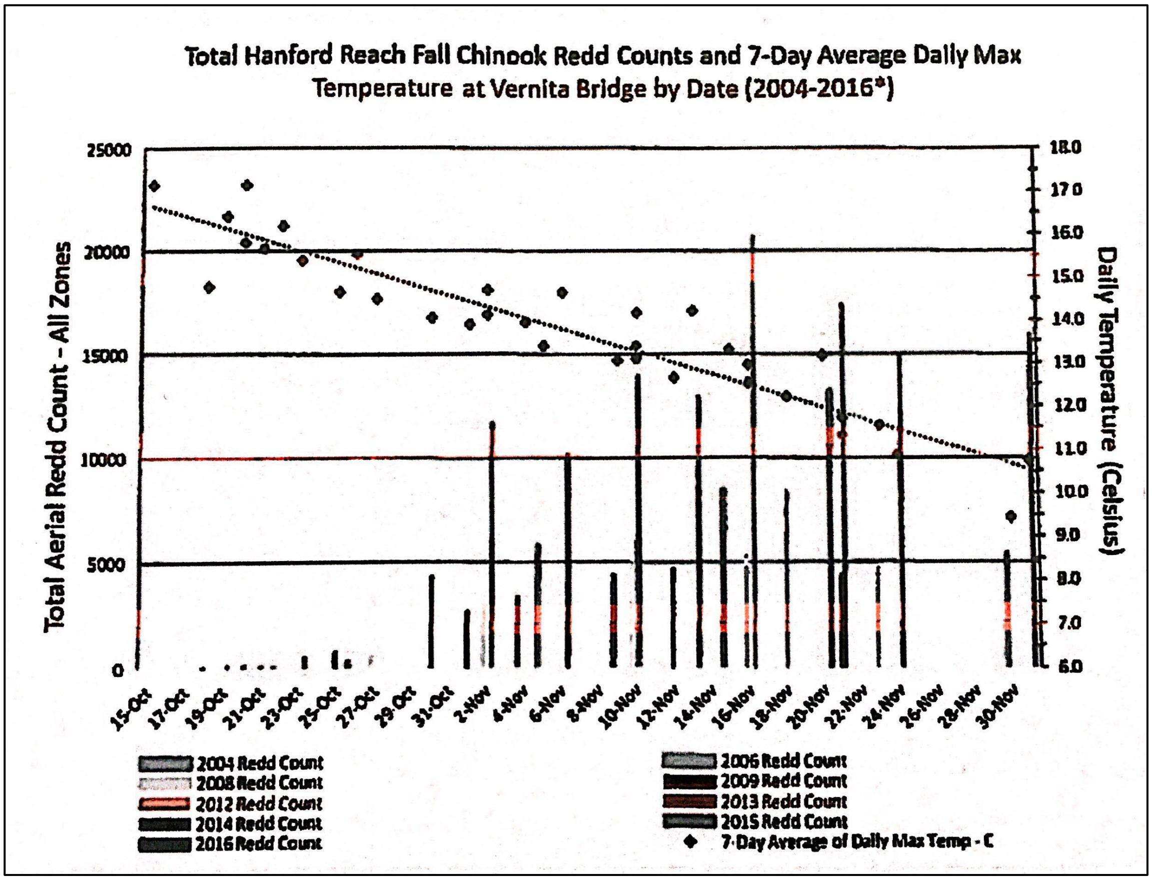

Figure 4.13. Nine years of total Chinook redd counts from all zones of the Hanford Reach between 2004 and 2016. Seven-day average temperature shown as points with regression line (Source: NOAA memo to USEPA; Appendix D).

4.6. Conclusions regarding alteration of thermal regimes

According to NOAA's Recovery Plan (2017), the operation of the Hells Canyon Complex contributes to a warmer thermal regime during the fall and may negatively affect the viability of adult Snake River fall Chinook salmon and egg survival, by causing some pre-spawning mortality and reducing egg viability or egg-to-fry survival.

4.7. Influence of Climate Change on water temperatures of the Action Area

In an assessment of stream temperature data using the NorWest statistical stream network model from more than 20,000 sites in the western U.S., Isaak et al. (2017) found that Pacific Northwest river and stream August mean temperatures have increased by an average of 0.17°C per decade (standard deviation = 0.067°C per decade) from the reconstructed trend over 40 years, from 1976 – 2015. For larger northwestern U.S. rivers, including Pacific Northwest rivers, estimated trends from time series at 391 sites across the northwestern U.S. revealed that warming trends are ubiquitous in the summer and fall months, with July – September mean river temperature increases of 0.18°C – 0.35°C per decade during 1996 – 2015 and 0.14°C – 0.27°C per decade during 1976 – 2015 (Isaak et al. 2018). They found that the average regional increase is largely linked to air temperature increases across the Pacific Northwest; however, at a local to sub-regional scale, other drivers, such as changes in discharge, can be influential.

5. EFFECTS OF THE ACTION

This section includes an analysis of the direct and indirect effects of the proposed action on the species and/or critical habitat and its interrelated and interdependent activities

5.1. The USEPA's Proposed Action on IDAPA 58-0102-1102 – analysis approach of the potential effects on threatened and endangered species

A direct effect is the direct or immediate effect of the project on a species or its designated critical habitat areas, whether beneficial or adverse. Direct effects result from the action and include the direct effects of interrelated actions and interdependent actions. Indirect effects are impacts that are caused by the action but occur later in time. This analysis of potential direct and indirect effects comprises likely effects that may be described qualitatively or quantitatively, depending upon the level of data and information available. The considerations in this analysis include:

1) The potential direct and indirect effects on threatened and endangered species from the Agency's proposed action include: (1) during the period and location of application of the SSC, including the time-period during which the criterion is changed; (2) the resulting effects during the transition period in temperature expected between Idaho's 19 daily average/22°C maximum criterion in Idaho and the SSC change from 13°C as a MWMT to 14.5°C as a WMT; and (3) the effects found for threatened and endangered species in the Idaho reach of the Snake River below its confluence with the Salmon River, in addition to species present above the Snake River's confluence with the Salmon River to Hells Canyon Dam. Idaho's SSC applies to the Snake River only to its confluence with the Salmon River, and the change from 13 to 14.5°C has potential implications for downstream threatened and endangered species.

2) The uncertainties associated with this review.

In the below review, in describing the potential effects of the Agency's action, the USEPA is analyzing the protectiveness of the 14.5°C SSC WMT. The USEPA's Temperature Guidance recommendation to protect salmon spawning is 13°C as a 7dadm.

5.2. Background on the EPA's Temperature Guidance and Best Available Science for Criteria Development

Idaho's criterion that is currently in effect for CWA purposes to protect fall Chinook spawning and egg incubation is 13°C as a maximum weekly maximum temperature (maximum of the 7-day average of the daily maxima). The basis for the 13°C criterion was derived from the USEPA's Northwest Temperature Criteria Project and resulting Temperature Guidance (EPA

2003). The USEPA's efforts to develop regional water temperature guidance to meet the specific needs of salmonids in Pacific Northwest streams and rivers, referred to as the Northwest Temperature Criteria Project, consisted of a multi-year collaborative process between states, tribes, and federal agencies to examine: (1) the most recent science on how temperature affects salmonid physiology and behavior; (2) the combined effects of temperature and other stressors on threatened fish stocks; (3) the pattern of temperature fluctuations in the natural environment; and (4) other issues relevant to developing temperature guidance to protect salmonids. After two rounds of public comments, the final guidance document, entitled USEPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards, was issued in April 2003 (USEPA 2003).

The scientific and technical foundation for USEPA's Temperature Guidance (USEPA 2003) was developed through an interagency technical workgroup, and included six, peer-reviewed, scientific papers provide the technical foundation for USEPA's final guidance document.

The six supporting papers, along with USEPA's final Temperature Guidance document (USEPA 2003), provide a comprehensive evaluation of the effects of temperature on salmonids in the Pacific Northwest and recommendations for optimal threshold temperatures that support these designated uses in State and Tribal water quality standards. Additional recent information considered in this document includes information provided in the Snake River Fall Chinook Recovery Plan, 2017, the 2015 NOAA and USFWS Biological Opinions on USEPA's Actions on Oregon's Temperature and Intergravel Dissolved Oxygen Water Quality Standards, which includes consultation on temperature water quality standards that apply to the Snake River below Hells Canyon Dam (a shared water with Oregon), the Idaho submission supporting documentation, a review of the Idaho submission by McCullough et al., 2013, and CRITFC (2019), the 2015 Pacific Salmon and Steelhead Recovery Report, and the results of a review of the scientific literature since 2015.

5.3. Rationale for the 13°C criterion

The diurnal variation when a 13°C criterion is applied is likely less than the diurnal variation in the summer, so USEPA hypothesize that a 13°C 7DADM criterion would result in maximum weekly mean between 10-12°C for a typical stream. The 13°C criterion is designed to protect spawning, egg incubation, and fry emergence for salmon and trout. Meeting this criterion at the onset of spawning for salmon and at the end of incubation for steelhead trout will likely provide protective temperatures for egg incubation [6 to 10°C (43 to 50°F)] that occurs over the winter (salmon) and spring (trout), assuming the typical annual thermal pattern. The 13°C criterion is designed to:
(1) protect ripe gametes inside adults during the weeks just prior to spawning [less than 13°C (55°F) constant],
(2) provide temperatures at which spawning is most frequently observed in the field [4 to 14°C (39 to 57°F) daily average], and

(3) provide protective temperatures for egg incubation [4 to 12°C (39 to 54°F) constant for good survival and 6 to 10°C (43 to 50°F) constant for optimal range] that occurs over the winter (salmon) and spring (trout), assuming the typical annual thermal pattern (EPA 2003).
Further, USEPA's Temperature Guidance identified the upper end of the optimal range for spring and fall Chinook spawning as 12.8°C (described in Issue Paper V), similar to USEPA's recommended criterion of 13°C as a 7dadm. Issue Paper V summarized the temperature literature for fall Chinook spawning as follows:
"Based on a survey of temperature effects on all aspects of spawning in fall-spawning salmonids, it appears that spawning temperatures in the spring and fall chinook spawning habitats having a 55°F (12.8°C) peak and then a declining trend would satisfy biological requirements.

Egg mortality, alevin development linked to thermal exposure of eggs in ripe females or newly deposited in gravel, and egg maturation are negatively affected by exposure to temperatures above approximately 54.5-57.2°F (12.5-14°C). Therefore, a spawning temperature range of 42-55°F (5.6-12.8°C) (maximum) appears to be a reasonable recommendation for Pacific salmon, unless colder thermal regimes are natural in any tributary."

The above literature survey compilation (EPA 2002, EPA 2003) included both the Olson and Foster (1957) and Olson et al. (1970) data and conclusions in identifying the ranges and recommendations for spawning criteria. In addition, the Temperature Guidance recommendations in general considered an NAS recommendation for adjustments to be added to excess mortality rates from temperature (LT50 after 7 days exposure), such that suggested protective criteria contain a buffer and result in LT1 rates of mortality for threatened and endangered species[6].

5.4. Hells Canyon Snake River Temperatures –Exposures Related to Agency's Action

Hells Canyon Snake River daily maximum temperatures can reach 23C in September and remain relatively high (often exceeding 19C) throughout October. Based on data collected through 2018 (pers comm. Idaho Power Company), September 1-October 22nd average daily maximum temperatures range from 19.5-20.6C, depending upon the year, while average daily maximum temperatures from October 9-October 22 range from 15.9-19.0°C (Table 5.1). There is a pattern of increasing Snake River temperatures over the years at this location, with 2010-2018 on average 1°C warmer than temperatures 1991-1999 (range 0.4-1.5C) (Figure 5.1).

[6] LT50: Lethal threshold concentration for 50 percent of the test organisms. National Academy of Sciences, 1972 cited in U.S. EPA. 2002. Issue Paper V: EPA Temperature Project.

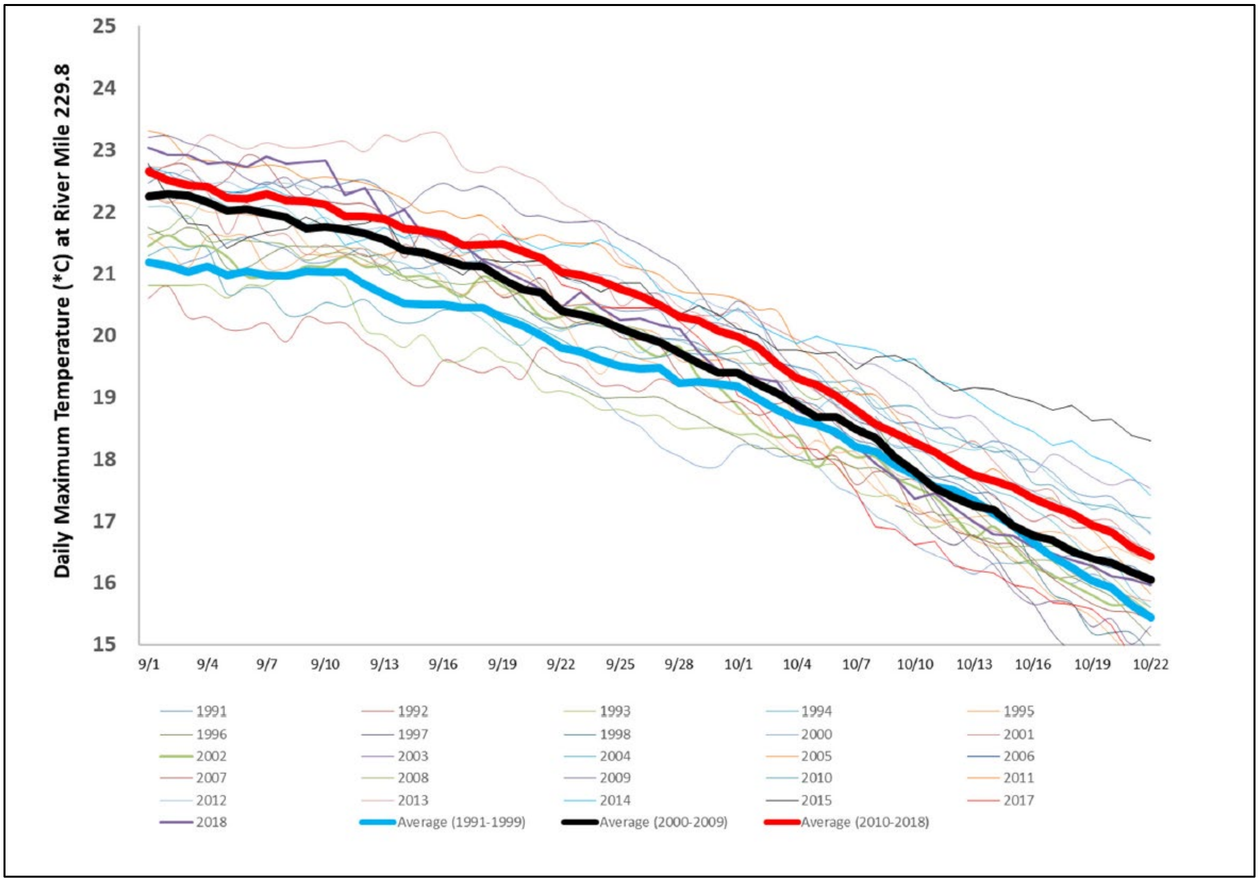

Figure 5.1. Compilation of daily maximum temperatures for the years 1991-2018 at RM 229.8. Thick lines represent means of three different decades. The average daily temperature decline at this site for both the October 9th through October 22nd and the October 9th through November 6th periods was 0.2°C (per day) [Memorandum from P. Leinenbach, Appendix B; using Idaho Power Company Data].

Table 5.1. Observed temperatures in fall transition (September 1- October 22) and early to end of October (covering early fall-run Chinook spawning period (October 9-22).

Table 1. Calculated Average Daily Maximum Snake River Temperatures During the Fall Period Observed at the River Mile 229.8 Monitoring Station During this Decade

	Assessment Period	
Year	September 1st through October 22nd	October 9th through October 22nd
2010	19.5	17.9
2011	20.4	17.4
2012	20.1	17.5
2013	20.4	16.7
2014	20.6	18.6
2015	20.5	19.0
2016	No Data	No Data
2017	Incomplete Data	15.9
2018	19.8	16.7

Under the proposed SSC, the first 7DADM averaging period is from October 23 through October 29 and must not exceed 14.5C. Assuming a 0.2°C decline during this period, daily maximum temperatures can be 14.8-15.4°C during the first four days of this period (Table 5.2). The proposed SSC is generally cooler than existing temperatures. The effects analysis below will focus on effects associated with changing the criterion from 13°C to 14.5C.

Table 5.2. Maximum allowed temperatures for the October 23-29 time-period; middle, new 14.5°C SSC, right, 13.0°C SSC. 13°C data provided for reference, only.

Date	Daily Max- 14.5°C SSC	Daily max- 13.0°C SSC
23-Oct	15.4	13.9
24-Oct	15.2	13.7
25-Oct	15.0	13.5
26-Oct	14.8	13.3
27-Oct	14.6	13.1
28-Oct	14.4	12.9
29-Oct	14.2	12.7
7-day Average Max	14.8	13.3

The anticipated limit on Snake River temperatures prior to October 23rd results from the connection between the applicable migration corridor criteria and the USEPA's proposed action on the spawning SSC. As described in its submission, IDEQ recognizes that there is a transitional decline in temperatures from Idaho's 22°C maximum/19°C daily average salmon migration criteria to the proposed 14.5°C SSC spawning criterion on October 23 (IDEQ 2012). Because USEPA's recommend 13°C spawning criteria is designed to provide protection for the two-week period prior to spawning to protect developing gametes, the USEPA is also evaluating the effects of the proposed SSC during this period. Given the mean 0.2C/day rate of decline (Figure 5.2; Appendix B; and IDEQ 2012), the 13°C criterion serves to effectively limit temperatures at or below 16.6°C during the pre-spawning and early spawning period, whereas, for the 14.5°C criterion, daily maximum temperatures can be as high as 18°C on October 9 and continue to exceed 16.5°C until October 17. See Figure 5.2 .

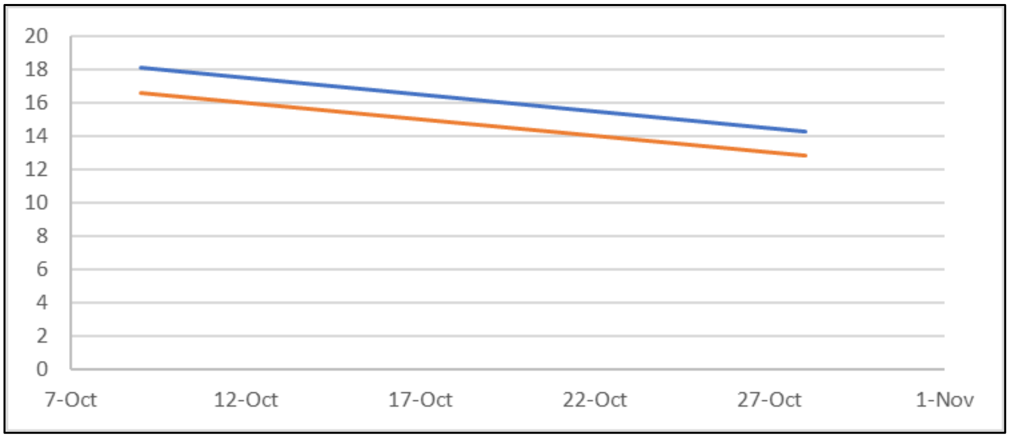

Figure 5.2. Comparison in daily temperature maxima with a 0.2°C rate of decline in Snake River temperatures between October 9 and October 28th in order to meet the SSC as a 7dadm on October 29; 13.0 (red line; provided for comparison) and 14.5°C (blue line).

5.4.1. <u>SR Adult Sockeye and Critical Habitat</u>

5.4.1.1. Presence in the Hells Canyon Reach of the Snake River

As established in Chapter 3, Snake River Sockeye and its critical habitat are present in this portion of the Snake River. However, adult SR Sockeye migrate directly up the Salmon River, and spawn primarily in lakes of the upper Salmon River watershed, therefore they are not likely to be directly affected in the Action Area.

5.4.1.2. Timing in the Hells Canyon Reach of the Snake River

At the time of year that the SSC applies, October 23-November 6, the majority of SR Adult Sockeye have completed their migration. According to the available data, SR sockeye complete their migration (Chapter 3, Figure 3.7, for 1994-2018 data) at Lower Granite Dam prior to the end of October in 19/24 years. The total number of adult SR Sockeye migrants was 17 in 2006, when adults continued to pass Lower Granite Dam until November 27. The estimated population exposed to elevated temperatures due to the Agency's Action is typically one individual in years when migration happened later, but in 2006 the number amounted to 5% of the SR Sockeye adults (Chapter 3, Figures therein). According to sockeye pit-tag data from the University of Washington DART database, 0.3% of migrants have been detected passing Lower Granite Dam after mid-October in the past 10 years (Figure 3.8).

5.4.1.3. Direct and Indirect Effects

As discussed above, Snake River temperatures in the two weeks prior to the start of the SSC and during the time-period of the SSC change would be warmer than river temperatures allowed with a 13°C criterion (Figure 5.2), given the presumed decline in river temperatures. Given that adult SR Sockeye are migrating up the Salmon River, the effects from this action are likely to be limited to the impact at the confluence with the Salmon, and downstream. As mentioned in (b), a subset of adult SR Sockeye is present at the time that the SSC change applies, with up to 5% of the run passing through from Lower Granite Dam during 2006 (Chapter 3, Figures therein). As a result of the SSC change from 13 to 14.5°C as a 7dadm, estimated Snake River temperatures from October 9 until November 6 would exceed the threshold for impacts to ripe gametes inside adults [less than 13°C (55°F) constant], in excess of 16°C during the two weeks prior to October 23. If sockeye were exposed these temperature conditions just prior to spawning there would be potential for adverse effects (e.g., Jeffries et al. 2012; Bowerman et al. 2016 (sockeye pre-spawning mortality rates of up to 80% based on Fraser River data). However, it is estimated that it takes approximately 40 days for adult SR Sockeye to travel from Lower Granite Dam to the Sawtooth spawning grounds in the upper Salmon River, most of which would be in the Salmon River, based upon the total distance (Crozier et al. 2014). Therefore, it is unlikely that a sockeye would be exposed to Snake River temperatures in the two weeks prior to spawning. Likewise, Salmon River temperatures are generally much colder than mainstem Snake River temperatures and serve as a refuge by cooling the Snake at the confluence (See Figure 4.4).

The most relevant of the critical habitat PBFs that could be affected by the Agency's action is water quality and specifically water temperature to protect adult Sockeye migration and spawning. Although migration corridor temperatures may be potentially affected by the Agency's action, the USEPA does not believe the Action will significantly adversely affect Sockeye critical habitat for the reasons described above.

Based upon the above review, the USEPA has determined that the Agency's action **may affect, but is not likely to adversely affect,** Snake River Sockeye and its Critical Habitat.

5.4.2. SR Adult Spring/Summer Chinook and Critical Habitat

5.4.2.1. Presence in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Spring/Summer Chinook and its critical habitat are present in this portion of the Snake River.

5.4.2.2. Timing in the Hells Canyon Reach of the Snake River and Potential Effects

At the time of year when Snake River temperatures may be affected by the Agency's action, October 23-November 6, adult Spring/Summer Chinook have already completed their migration (Chapter 3.5), with 100% of adults crossing Lower Granite Dam prior to August 1 (based upon 1994-2018 data)[7]. While juveniles outmigrating and rearing may be present at this time of year, the USEPA's Temperature Guidance (2003) recommended a criterion of 18°C for combined migration and rearing, and 16°C to protect core rearing, as 7dadms, both of which are higher than the 14.5°C SSC.

The most relevant of the critical habitat PBFs that could be affected by the Agency's action is water quality and specifically water temperature to protect adult Spring/Summer Chinook migration, spawning, and juvenile migration and rearing. Although water temperatures may be potentially affected by the Agency's action, the USEPA does not believe the Action will

[7] Columbia River DART; downloaded February, 2019.

significantly adversely affect Spring/Summer Chinook critical habitat for the reasons described above.

Based upon the above review, the USEPA has determined that the Agency's action **may affect but is not likely to adversely affect**, Snake River Spring/Summer Chinook and its Critical Habitat.

5.4.3. SR Steelhead Migration and Critical Habitat

5.4.3.1. Presence in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Steelhead are an ESU present in the Snake River.

5.4.3.2. Timing in the Hells Canyon Reach of the Snake River

At the time of year of the SSC, October 23-November 6, SR adult Steelhead, during most years of record, a subset of the population (typically 10- 25%) crosses Lower Granite Dam just prior to or during the time-period of the SSC change. Juveniles may be present at this time of year.

5.4.3.3. Direct and Indirect Effects

As established in Chapters 3 and 4, Snake River Steelhead start to experience acute adverse effects at river temperatures at or above 19C, when they seek out cold water refuges and other means of maintaining a lower body temperature (Goniea et al. 2006; Keefer et al. 2018; Chapter 4, Figure 4.3). Because of the SSC change from 13 to 14.5°C as a 7dadm, Snake River temperatures from October 9 until November 6 would be allowed to exceed the threshold for impacts to ripe gametes inside adults [less than 13°C (55°F) constant] for a longer period of time than with the prior (13C) criterion. However, the stream type, Type "A", steelhead found in the Snake River Basin hold in larger rivers over winter to mature and typically do not spawn until the following March after dispersing up to headwater spawning grounds (Chapter 3.7). Although potential coldwater refuges have been identified (see Chapter 4) below and within the action area, data on the sufficiency of these refuges and whether they have served historically as refuges for steelhead is currently lacking. Likewise, there is uncertainty in the exposure timeframe of adult SR steelhead in the Snake River during their migration– the USEPA has not identified the time of travel for adult SR steelhead within the Snake River until they reach their spawning grounds, and whether adult SR steelhead are exposed during the full time-period of change of the criterion, or a subset of time that the criterion change applies or would be expected to affect temperatures in the Hells Canyon Reach of the Snake River. It is unclear whether sufficiently distributed coldwater refuges are available to mitigate any effects to adult SR steelhead from the change in the SSC, since although the change in the SSC to a magnitude of 14.5°C is still well below the 19°C threshold for heavy refuge use, the SSC allows for warmer temperatures earlier in the migration season (Figure 5.2). However, because gamete formation and spawning take place several months after the change in the SSC applies (Chapter 3), and because 13°C is retained as the spawning criterion from November 7-April 15, the USEPA finds that exposure to higher Snake River temperatures due to the Agency's Action is unlikely to affect steelhead gametes and spawning.

Likewise, while juveniles outmigrating or rearing may be present at this time of year, the USEPA's Temperature Guidance (2003) recommends a criterion of 18°C as a 7dadm to protect juvenile migration and rearing, and 16°C to protect core juvenile rearing, both of which are higher than the criterion that is the subject of the Agency's Action, 14.5C.

The relevant critical habitat PBF that may be affected by the Agency's action is water quality and specifically water temperature for adult upstream migration, spawning, and juvenile migration and rearing. For the reasons described above, the Agency's action is not likely to significantly affect water quality that impacts these steelhead lifestages.

For the above reasons, the EPA has concluded that the Agency's Action **may affect but is unlikely to adversely affect**, Snake River Steelhead and its critical habitat.

5.4.4. Fall Chinook

5.4.4.1. Fall Chinook spawning through fry emergence

(a) Presence in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Fall Chinook are an ESU present in the Snake River.

(b) Timing in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Fall Chinook are present in the Snake River during the time-period affected by the Agency's action, as migrating, holding, and spawning adults through redd emplacement, and egg fertilization and incubation.

Direct Effects: Lethal and sublethal impacts to eggs and fry

Due to its strong influence on metabolic efficiency and senescence, the thermal regime that migratory fish, and eggs and fry experience establishes the lower bound for reproductive success for fall Chinook (Plumb et al. 2018). Laboratory studies have shown that fall Chinook initial spawning temperatures greater than 14.8 °C result in substantially increased levels of egg mortality [Seymour 1956; Olson et al. 1970; Geist et al. 2006. In its submission, in combining three studies together with a spline regression model, IDEQ identified a 95% confidence interval lower bound confidence interval of 15.3°C for an increase in excess mortality (IDEQ, 2012). However, a more gradual range of thresholds for increasing egg and fry mortality was identified in Olson et al. 1970; Table 5.3.

Table 5.3 Threshold temperatures for fall Chinook eggs and fry mortality in a declining thermal regime, October 30 cohort (Source: Olson et al. 1970).

	54.6F/12.6C	56.6F/13.7C	58.6F/14.8C	60.6F/15.89C	62.6F/17C	64.6F/18.11C	66.6F/19.22C
Egg+fry mortality gross	4.57%	3.64%	11.01%	28.14%	59.55%	97.43%	100%
Net- excess background control	0	-0.93%	6.44%	23.57%	54.98%	92.86%	95.43%

For the Geist et al. 2006 study, the adult fall Chinook used in the experiments did not undergo the complete lengthy migration at high temperatures that Snake River fall Chinook experience, and in addition, the fall Chinook parents were held at 12 °C (54 °F) prior to spawning, a temperature that is considerably cooler than that observed in the Upper Hells Canyon reach prior to spawning. Such holding temperatures are protective of gamete formation, which is adversely affected above 13°C. Pre-spawning mortality, and damage to gametes and their influence on egg development to the fry stage, diversity, and total spawn count impacts comparable to what fall Chinook would experience in the Snake River were therefore not reflected in the Geist et al. 2006 study[8][9]. Additionally, the Geist et al. 2006 study was not a controlled study; no control egg population was held at the 12°C holding temperature and, therefore, it is difficult to accurately assess the excess mortality rate associated with the treatments in Geist et al. 2006. Olson et al. 1970 exposed gametes from only 2 adult fish, which could limit the application of such a narrow dataset to a broader natural population, and the exposure temperatures were ambient conditions that may not reflect the increase in temperatures throughout the Snake and Columbia River systems since the 1960s (Figure 5.1, Chapter 6). In their experiments, Olson et al. (1970)

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[8] NOAA (2015), analyzed the implications of 13C or alternatively a warmer criterion based on the results of Geist et al. 2006, as follows:
"Richter and Kolmes (2005, p. 38) confirmed the conclusion of EPA (2003) that a 13.0°C criterion as a 7DADM is adequate to protect spawning and incubation in Chinook salmon, noting that it is "consistent with the upper temperature range for optimum survival of chinook [sic] salmon embryos and alevins and [is] within reported temperature ranges for successful spawning." The study by Geist et al. (2006) described in the IGDO discussion above included information on the effects of water temperature on fall Chinook salmon in the laboratory (Geist et al. 2006). Fall Chinook salmon embryo survival from fertilization to hatch and from fertilization to emergence was lower at 13.0° with DO at saturation than it was for some of the temperature/dissolved oxygen combinations with higher temperatures and moderate to high (but below saturation) DO concentrations. We view these temperature results with caution, because the authors held the pre-spawn adult salmon at a constant water temperature of 12° C, which is colder than the river during spawning. This may have protected gametes in the holding fish from injury and improved the later survival to emergence in some of the warmer treatments."

[9] Conclusion by Geist et al. 2006 included a caveat that it is possible that the cooler pre-spawning temperatures led to higher survival rates than those found in nature

followed eggs through the fry development stage and found that all but the two coldest temperatures of exposure led to significant levels of increased mortality (Olson et al. 1970).

The increase from the current criterion of 13°C to 14.5°C as a 7dadm, together with Idaho's use of an additional 0.3°C de minimis allowance[10], would allow maximum river temperatures to legally reach 15.4°C as a maximum (with daily maxima ranging between 14.2 and 15.4°C given the typical decline in temperatures in the Snake River at this time of year) within the period of October 23-October 29 (Table 5.2), identified in the recovery plan as within the range of "unknown effects", and on some days potentially above the 95% confidence interval lower bound for the transition point to excess mortality (15.3°C)[11] identified by IDEQ and Idaho Power Company in the submission[12]. A 14.5°C 7dadm criterion would allow temperatures to exceed 15.4°C throughout the weeks prior to October 23rd. Likewise, for eggs deposited prior to October 29 when allowable temperatures are at or above 14.8C, significant excess mortality is likely to result (6.44-23.57% excess mortality per the most comparable October 30th cohort of Olson et al. 1970). The Olson et al. (1970) study looked at 4 incubation dates, October 30th, November 14th, November 23rd, and December 8th. The October 30th data are most relevant to the time-period of interest for this SSC (from October 23rd through November 6th). For the October 30th-spawned lots, there was no excess total (egg + fish) mortality found at initial spawning temperatures of up to 13.7C, but excess mortality at 14.8°C was 6.44% and excess mortality at 15.9°C was 23.57%. When initial exposure temperatures reached 17°C and above, mortality ranged from 54.98% to greater than 90% (see Table 5.3). The Olson and Foster 1957 study results were variable, with the control (13.8°C) resulting in 16.1% total mortality; 11.6°C resulting in a reduced mortality of 7.8% total mortality; total mortalities of 10.1% and 10.4% for 15 and 16°C, respectively; and much greater total mortalities of 79% for 18.4°C.

Field and modeling data: Temperature data during weekly spawning surveys in this reach of the Snake River from 2000 through 2009 show that while redd emplacement occurs at maximum weekly maximum water temperatures in excess of 13 °C (55 °F), only a small percentage of all fall Chinook salmon redd emplacement occurred in the reach when water temperatures were

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[10] IDEQ has adopted a de minimis provision allowing 0.3C above the applicable criteria into state law and the EPA anticipates that the provision will be submitted by Idaho to the EPA for its review and action under Section 303(c) of the CWA in spring 2019

[11] Due to differences in methodologies, exposure temperatures, exposure conditions, and rates of decline, the EPA finds that it may be inappropriate to combine the data from these three studies and apply a regression, without normalization efforts. Additionally, in the regression a datapoint from the Olson et al. 1970 study was not converted properly from Fahrenheit to Celsius. As summarized in the Idaho submission: "The Geist et al. (2006) study used a 0.2°C daily rate of decline, which was comparable to data from the Snake River, whereas the Olson and Foster (1957) study used a daily rate of decline of 0.18°C. The Olson et al. (1970) study had a more variable rate of decline ranging from 1.1°C/d to 1.7°C/d (estimated from figures in the report) because they used Columbia River water at the existing temperatures as the baseline. The two Hanford Reach studies used Columbia River water, whereas the Geist et al. (2006) study used well water. The Hanford Reach studies monitored survival to a point past emergence whereas the Geist et al. (2006) monitored survival to emergence. Olson et al. (1970) was replicated over four spawning dates, whereas the Olson and Foster (1957) and the Geist et al. (2006) was conducted using one spawning date. These differences may be factors in the slightly higher threshold reported by Geist et al. (2006) than observed by Olson and Foster (1957)." Due to these differences in methodology and associated uncertainties with the regression model, the EPA has analyzed the effects based upon the individual study results here, along with other pertinent data.

[12] IDEQ 2012.

above 14.5°C as a 7dadm, and very few when river temperatures reached 16.5 °C (61.7 °F) or higher (NOAA 2017). The weekly spawning surveys conducted by the Idaho Power Company, U.S. Fish and Wildlife Service, and others from 1994-2017 indicate that the median initiation date that Snake River fall Chinook salmon in the Upper Hells Canyon reach emplace redds is October 22 when water temperatures typically exceed 14.5 °C, through November 20, when water temperatures drop to about 12 °C (54 °F) (unpublished data provided by Brett Dumas, Idaho Power Corporation to Rochelle Labiosa; comparable to 2000-2009 pattern provided in NOAA 2017). During a 13-year study period (1991-2003), 4 percent of redds surveyed were initiated when water temperatures were greater than 16.5 °C (61.7 °F) (NOAA 2017). Typically, 10 to 20 percent of redds are deposited between October 23 and 31, when water temperatures are 14.5 to 16 °C (58 to 61 °F) (NOAA 2017).

There is much uncertainty around the appropriate threshold temperature to prevent significant mortality and/or compromised development in eggs and fry. As previously mentioned, the USEPA's Temperature Guidance recommended 13°C as a 7dadm for the spawning season, derived from a wide array of literature primarily including laboratory experiments under constant temperature exposures across different salmon species. According to NOAA's 2017 fall Chinook Recovery Plan[13], fall Chinook salmon that enter freshwater in the summer and fall, may emplace redds in warmer water than coldwater fish that enter freshwater in the spring, such as spring/summer Chinook salmon. However, according to recent modeling by Connor et al., 2018, there is a high probability that a significant proportion of the earliest emplaced redds are empty or eggs are unable to develop, due to energy deficits leading to pre-spawning mortality or premature spawning. Similarly, while adult returns from 2011 through 2015 (2009-2013 estimated ocean outmigration year) were robust, driven by exceedingly warm and food-poor conditions in the ocean (Peterson et al. 2018) from 2015 (2013 outmigration year) onward, adult returns have steeply declined (Figure 3.2). Poor ocean conditions and higher temperatures in migration corridor streams can lead to adult migratory salmonids in general in poorer condition upon returning to natal streams, and a lack of genetic diversity could weaken the resilience of the species when environmental conditions change (see e.g., Crozier et al. 2008). For Snake River fall Chinook, the extirpation of certain historical populations, as well as hatchery influence (Chapter 3), may have affected genetic diversity of this ESU (Chapter 3) (NOAA 2017). Similarly, for fall-run Chinook, the 2019 run forecast is predicted to be poor, less than 50% of the 10-year average (Figure 5.3; NOAA 2019). Therefore, the line of evidence that fall Chinook are robust and produce redds and eggs at warmer temperatures equivalent to current conditions should be interpreted cautiously in light of new information, in particular, given 1) the recent declines in adult returns and 2) the modeled energy losses for fall-run Chinook provided in Connor et al. 2018 that indicate that early redd emplacement under currently warmer conditions is not necessarily an indicator that eggs and fry are produced, with high rates of unsuccessful spawning expected given the poor energy status of fall Chinook just prior to spawning under such conditions (Figure 5.5 through Figure 5.8). For unsuccessful spawners, energetic modeling which identified "unsuccessful spawners" projected pre-spawning mortality to be 20% of the total mortality, with the majority to be "premature spawners" or spawners that build redds but die prior to successful spawning (Figure 5.5). Pre-spawning mortality based on the 6 available datasets for the Upper Hells Canyon Reach are compiled in Bowerman et al. 2016 (Figure 5.6) estimate and average overall pre-spawning mortality of 5%, maximum of 10%. The impact is

[13] NOAA 2017. Snake River Fall Chinook Recovery Plan.

greatest for early spawners (prior to 10/28) according to Connor et al. 2018 modeling; 80-100% of spawners are identified as pre-spawning mortality during that early time-period for the range of conditions studied, which included a range of river temperature patterns (Figure 5.7; Appendix B). Likewise, in Figure B, according to Connor et al. 2018, high levels of pre-spawning mortality and premature spawning are found in the early spawning period, and typically comprise all spawners in the early spawning period prior to the end of October (Figure 5.8).

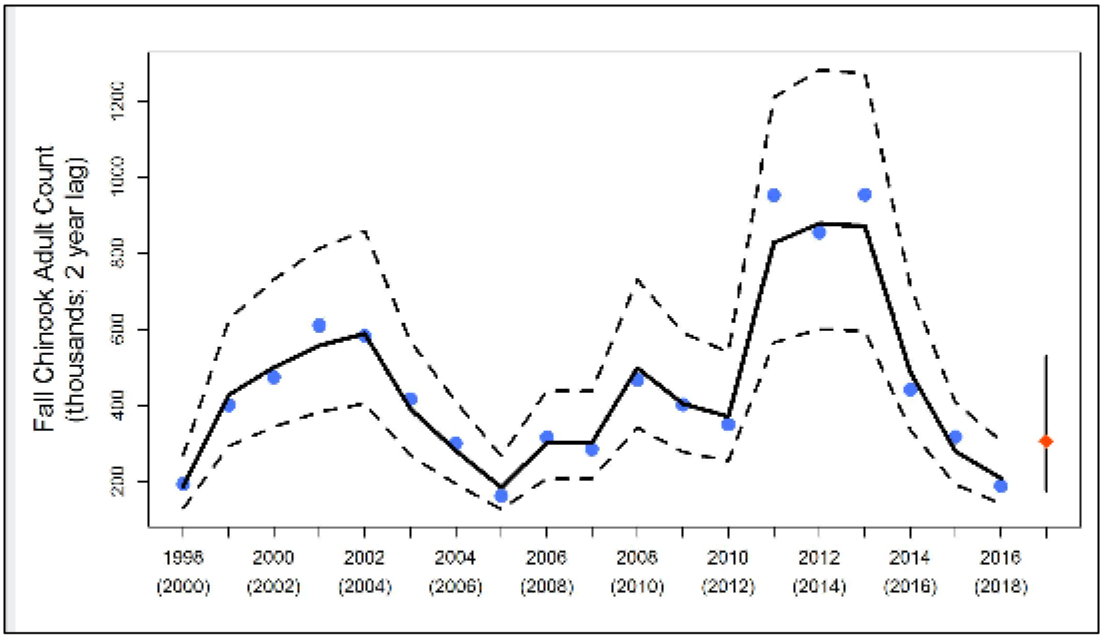


Figure 5.3. 2019 forecast for fall Chinook adult returns at Bonneville (red dot with uncertainty band) compared to previous annual counts (blue dots with CIs). (excerpted from https://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/g-forecast.cfm March 7, 2019)

The USEPA's action would allow temperatures prior to October 23 to legally range up to 20°C as a 7dadm so long as sufficient coldwater refugia are present, or 19°C as a daily average and 22°C as a maximum to protect cold water fish and salmon and steelhead migration, whichever is more stringent (See Water Quality Standards, Section 2). Exposures to these migration corridor criteria, would result in 100% mortality to eggs and fry. Given the typical rates of decline in the Snake River, the USEPA understands that the migration corridor criteria are meant to be implemented as summer maxima, and therefore effects will be lessened by a gradual decline of up to 0.2°C per day between the migration corridor criteria in summer and the spawning criteria in later fall (IDEQ 2012). As mentioned in the Recovery Plan (2017), approximately 4% of redds (NOAA 2017) are typically emplaced prior to the time that the salmonid spawning criterion would be applicable, and eggs and fry located in these redds would likely be reduced by 55-98% mortality, per the Olson and Foster 1957, Olson et al. 1970 and Geist et al. 2006 estimates, for threshold temperatures in excess of 17C. This loss percentage is greater than 50% of the tail of the distribution for a listed species, which could impact diversity of fall Chinook (see e.g., EPA 1985). Connor et al. 2018 further interpreted the Geist et al. 2006 results, estimating that embryo survival decreased by 10.9 percentage points for every 0.1°C increase in DD>16.5. Based upon the available fall Chinook spawning and egg incubation/development studies in declining thermal regimes, an additional 2.4-4% of eggs could be lost for this time-period prior to the SSC application start date on 10/23. With the current proposed 14.5°C SSC there can be 100% excess egg and fry loss rate for this time-period (Figure 5.2).

As shown in Table 5.2, due to the averaging period and time-period of application of the SSC, up to a 15.4°C maximum temperature is allowed on October 23 by the 14.5°C 7dadm criterion, given the typical 0.2°C daily decline in temperatures at this time of year. As previously mentioned (Table 5.3), according to Olson et al. (1970), increased excess mortality to eggs and fry over a comparable suite of temperatures ranged from 6.44% (at 14.8°C) to 23.57% (at 15.9°C). Because the temperature criterion is presumed to be gradually declining over the period of application, in order to meet 13°C as a 7dadm on 11/7, the time-period where temperatures may cause excess mortality are in the first week of application of the criterion, and prior to that time. The percentage of redds emplaced October 23 to October 31 is roughly 10-20% (NOAA 2017) but when the peak redd emplacement is earlier in the year (Connor et al. 2017), total cumulative percentages of shallow water (<3m; typically 2/3 of the total redds) Snake River redds emplaced prior to 10/29 can be in excess of 30% (Connor et al. 2017, attached as Appendix E). The USEPA estimates that the excess loss of eggs due to the change in the SSC to 14.5C for the timeperiod from 10/23-10/29 based on the higher bound 23.57% loss rate and the 39.3% upper bound percentage of surface redds (Connor et al. 2017) (39.3% of redds prior to 10/29- 4% of estimated early redds prior to 10/23=35.3% of redds 10/23-10/29), results in up to 8.3% of eggs and fry lost to mortality[14] from 10/23-10/29 which, together with the 4% additional loss prior to 10/23 results in a total potential loss of eggs and fry of 12.3%. An estimation using the lower estimated excess mortality rate at 14.8C of 6.44% (Olson et al. 1970), would result in a total of 2.3% of mortality for eggs and fry during the 10/23-10/29 period, which, together with the additional 4% of early redds lost prior to 10/23, would result in a lower bound total egg and fry mortality rate of 6.3% of eggs and fry.

[14] Note that because Connor et al. 2017 reported spawning dates as "flight date – 7," the spawning date recorded as e.g., 10/29 encompasses redds from 10/29-~11/4. Several spawning dates reported on or around 10/29 reflected redd percentages much higher than estimates from 10/23-10/29 (reported as 10/23 in Connor et al.); for 10/29-11/4, ranging up to 76% of redds.

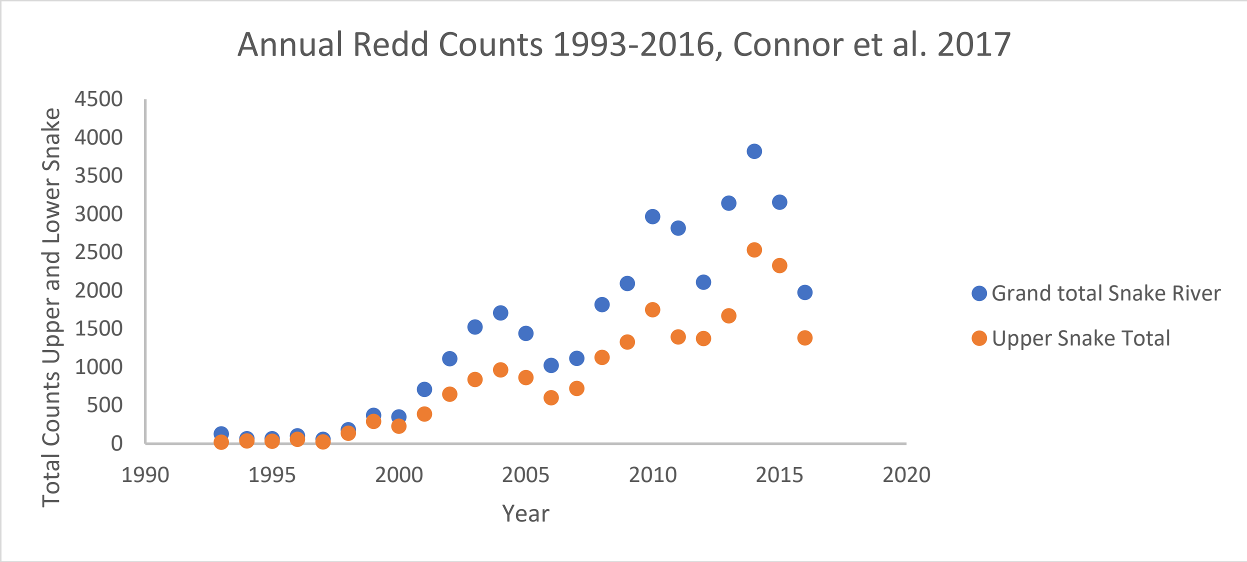


Figure 5.4 (a). Grand total of all redds for Snake River (blue dots) and total redds for the Upper Snake (orange dots) each year, 1993-2016 (Connor et al. 2017).

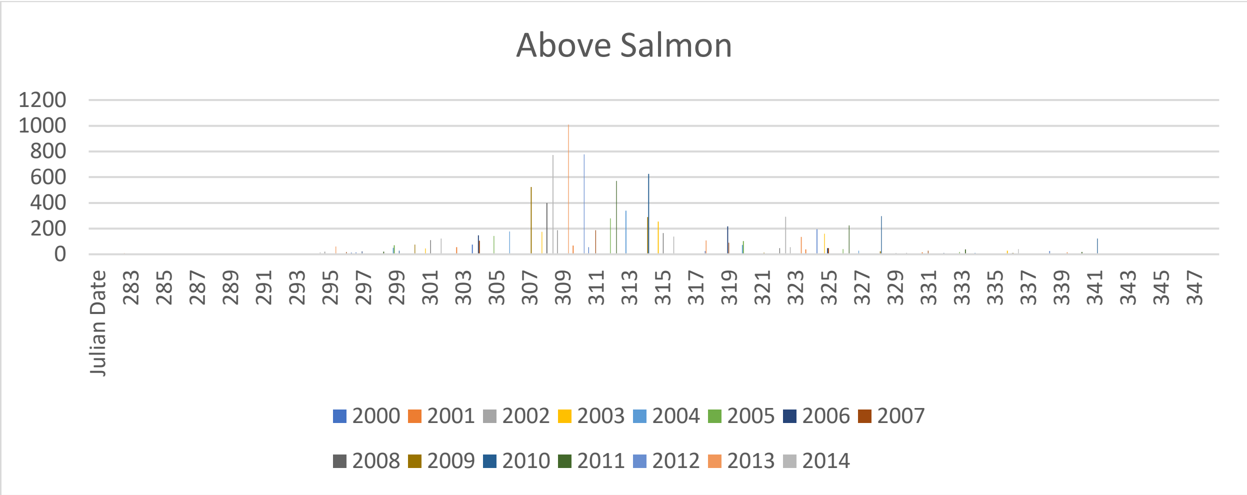


Figure 5.4 (b). Redd Distribution by Julian Date and Year for the Snake River above the Salmon River. October 23 is 296 (or 297 if leap year). (Source: Idaho Power Company, unpublished data pers. comm. with R. Labiosa, 2018; note that total counts and distribution differ from those in Connor et al. 2017; Idaho Power includes aerial surveys only).

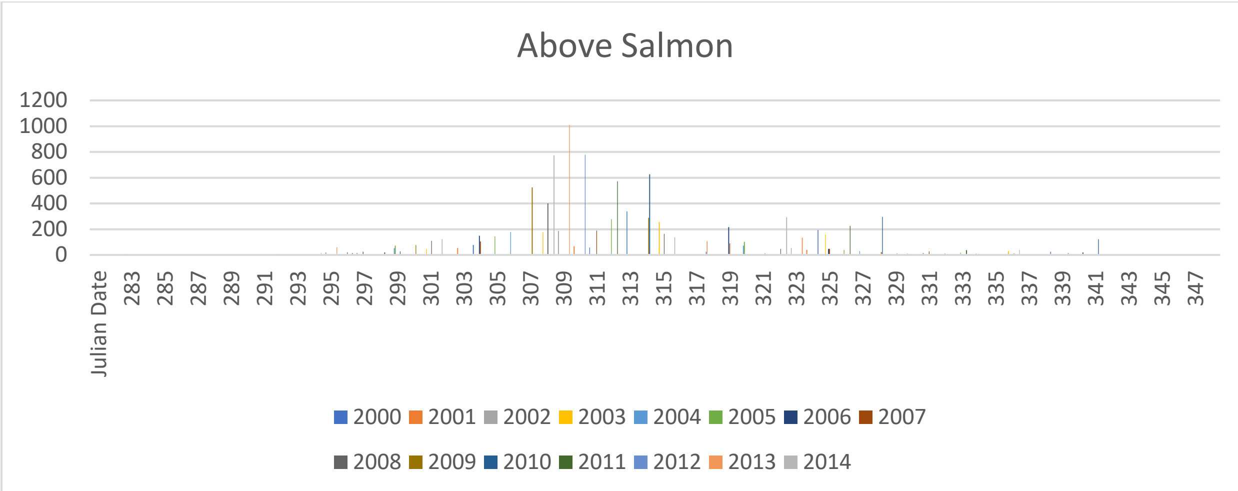


Figure 5.4 (c). Redd Distribution by Julian Date and Year for the Snake River below the Salmon River. October 23 is 296 (or 297 if leap year). (Source: Idaho Power Company, unpublished data pers. comm. with R. Labiosa, 2018; note that total counts and distribution differ from those in Connor et al. 2017; Idaho Power aerial surveys only).

Based on the above, the Agency' Action is estimated to result in an upper bound of potential total annual loss of eggs and fry due to excess lethality and sublethal impacts for spawned eggs of 12.3% (6.3% at the lower 14.8°C mortality rate). Note in most years current Snake River Hells Canyon Reach temperatures are well above the temperatures allowed by the 14.5°C SSC and therefore, the currently level of mortality to eggs and fry is expected to be higher in most years than what the 14.5°C SSC allows (see e.g., Appendix A Table 3 and Table 5.4).

Table 5.4. Snake River Hells Canyon 7dadm temperatures measured at the penstock monitoring location showing exceedences of 13.3°C, 1991-2017. (Source: Idaho Power Company 401 certification proposal (2018)—Table 6.1-4).

Year	7DAM Temperature (°C)	Criteria Exceedance (°C)	Duration (days after 10/29)	Cumulative Thermal Load Exceedance (bkcal)	Annual Average Flow (cfs)	Water-Year Category
1991	16.4	3.1	12	453.2	10,400	Low
1992	15.8	2.5	16	551.0	8,400	Low
1993	15.7	2.4	10	366.5	16,500	Medium
1994	15.5	2.2	12	353.0	10,800	Low
1995	14.6	1.3	7	114.8	17,500	Medium
1996	14.8	1.5	8	150.2	24,600	High
1997	13.3	0.0	0	0.0	32,000	High
1998	14.0	0.7	6	58.7	23,000	High
1999	14.5	1.2	8	181.4	22,900	High
2000	15.0	1.7	9	192.9	15,100	Medium
2001	15.8	2.5	14	422.2	9,800	Low
2002	15.3	2.0	8	210.3	11,000	Low
2003	16.8	3.5	13	547.7	11,700	Low
2004	16.3	3.0	15	500.4	10,900	Low
2005	15.7	2.4	15	456.0	11,100	Low
2006	15.3	2.0	8	184.9	21,500	Medium–high
2007	14.5	1.2	9	116.3	11,000	Low
2008	14.9	1.6	10	175.1	12,700	Low
2009	14.6	1.3	6	95.2	14,400	Medium–low
2010	16.8	3.5	20	809.9	13,300	Medium–low
2011	15.4	2.1	11	428.0	24,900	High
2012	15.8	2.5	16	438.1	15,800	Medium
2013	15.3	2.0	11	277.4	9,700	Low
2014	17.2	3.9	21	1,044.9	11,200	Low
2015	17.9	4.6	23	1,256.4	10,200	Low
2016	16.0	2.7	24	686.5	12,200	Low
2017	14.4	1.1	8	105.9	25,600	High

Note: 1993 AND 2001 temperature data was not collected at the penstock monitor, so data collected within 20 miles downstream of HCD was used to fill the data gaps so a representative cumulative thermal load exceedance could be calculated for those years.

Direct Effects: Impacts to gametes and embryo loss

Adverse holding temperatures for adults also can lead to later excess egg mortality (Jensen et al. 2006). In a comparison between holding temperatures, a colder site with mean daily temperatures below 9°C resulted in 3.04% mortality for eggs, in comparison to 11.8-13.4% mortality for eggs at warmer locations with maximum mean daily temperatures ranging from approximately 15.5-23.5°C during the holding period [note that many females died prior to spawning after transfer into the warmer ambient locations; reported estimates are for successfully spawned eggs, only]. Although the 13°C criterion allows for temperatures to exceed 15.5°C in the early spawning period prior to October 23, the number of degree days over 15.5 is reduced by approximately two weeks in comparison to the 14.5°C criterion, given the typical rates of decline in temperatures at this time of year (Figure 5.4 (a)). An additional 8.8-10.4% of excess egg loss is likely to occur due to impacts to gametes from sustained excess temperatures for over two weeks prior to spawning. This range comports well with the simulations of Connor et al. 2018 (estimated embryo loss for successful spawners under current conditions in the Snake River of 5-20%, for October 27-November 1, depending upon the year, 2010-2015), and an approximate midpoint of 10% (Figure 5.10).

In summary, there are likely adverse effects for fall Chinook eggs from the Agency's action, stemming from both egg death due to exposure to higher than optimal temperatures, and loss of gametes due to cumulative exposures to high temperatures during gamete formation prior to spawning.

Direct Effects: Eggs and fry located below the confluence of the Salmon River not protected by an Idaho spawning criterion

The Upper Hells Canyon Contiguous Reach from the Hells Canyon Dam at rkm 398.7 to the Salmon River at rkm 302.9 (95.8 rkm total), and the Lower Hells Canyon Reach, from rkm 302.9 to rkm 234.0 (68.9 rkm total) are identified as two out of three primary spawning areas for Snake River fall Chinook, with the other primary area being the Clearwater River (Connor et al. 2018). The 13°C SSC that is currently effective in Idaho to protect fall Chinook spawning applies only to the confluence with the Salmon River. However, redds are emplaced in the Snake River below its confluence with the Salmon River in a second primary area below the Salmon River (Chapter 3, Figure 3.4, Since 2001 from 30 to 50% of redds are emplaced below the Salmon River confluence, depending on the year). The USEPA does not have information on the number of redds located between the confluence with the Salmon River and the Idaho, Oregon and Washington border (12.3 miles of the Snake River); Oregon applies its spawning criteria of 13°C as a 7dadm above and below the confluences with the Hells Canyon Dam, to its border with Washington (See Chapter 6). The USEPA's proposed action and change in the spawning criterion from 13°C to 14.5°C will allow the propagation of warmer river temperatures downstream, with adverse effects as described above for the portion of the Hells Canyon Reach above the confluence with the Salmon River. Without a cap on temperatures, anthropogenic sources could result in water temperatures up to 17.5°C (WA WQS) as a 7dadm or 19 daily average and 22°C maximum (ID WQS), whichever is more stringent, downstream of the area of complete mix from the upstream waters maintained at the spawning criterion, during the full spawning period. Although the Salmon River (and further downstream, the Clearwater River) cool the Snake River below the Salmon River (Chapter 4, Figure 4.3), Snake River maximum temperatures (Figure 3.5) typically exceed 17°C as a 7dadm prior to October 23. Effective implementation of downstream protection provisions and existing use (antidegradation) protective provisions may reduce these potential effects (See Chapter 2).

5.4.4.2. Adult pre-spawning impacts

(a) Presence in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Fall Chinook and its Critical Habitat are present in the Snake River.

(b) Timing in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Snake River Fall Chinook are present at the time of year of the SSC in the Snake River, as migrating, holding, and spawning adults and eggs.

5.4.4.3. Direct and indirect effects

There is information in the Snake River Fall Chinook Recovery Plan (NOAA, 2017) that identifies the September to early October time-period as a critical time-period for potential pre-spawning mortality, noting that "Nonetheless, in some years, adults passing Lower Granite Dam in late August and early September may still be exposed to 18 to 22 °C (64 to 72 °F) water temperatures for several days or weeks prior to spawning in this reach, and the prolonged exposure of adults to elevated temperatures in the migration corridor and spawning areas could potentially result in reduced spawning success and some egg and fry mortality…" National Oceanographic and Atmospheric Administration (NOAA) (2017) also speculate that cold water refuges may mitigate some of the thermal impacts from holding in Snake River temperatures in excess of thermal tolerances, including at the confluences with the Clearwater and Salmon Rivers (Mann and Peery 2005; Jensen et al. 2005, 2006) and potentially other "small tributaries" however, few details are provided on the sufficiency of refuges to offset the thermal exposures to adults. Further, use of refuges can be disadvantageous in terms of energy consumption (Plumb 2018).

In addition, as discussed in Connor et al. (2018), the literature including Bowerman et al. (2016) comprise more fine-tuned estimates of pre-spawning mortality for the genus *Oncorhynchus*, based on the definition that it is the mortality observed after the adults reach the spawning grounds but before eggs and milt are successfully released. Bowerman et al. (2016) found that pre-spawning mortality increased as the time in freshwater and distance migrated to the spawning grounds increased. This indicates that there is a link between energy expenditure over the migration and holding time and travelling distance which can lead to death.

As shown in Figure 5.7, Upper Hells Canyon Reach (above the Salmon River) fall Chinook pre-spawning mortality estimates based on simulations of energy loss for prior to and during the time the SSC applies are a high percentage of all spawning adults, ranging from 80-100% through October 29 in all years, with lesser degrees of pre-spawning mortality, depending upon the year, through early November (Connor et al. 2018). Additional estimates of pre-spawning and unsuccessful spawning populations and effects on embryos are presented in Figure 5.8, Figure 5.9, and Figure 5.10.

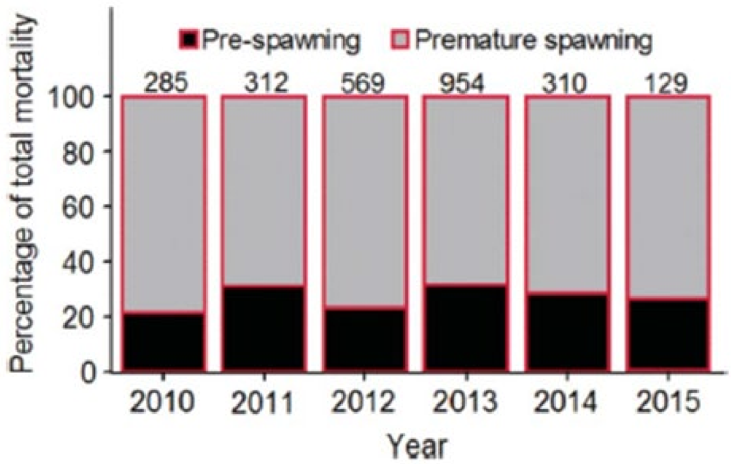


Figure 29. The percentages of the PIT-tagged, hatchery-origin fall Chinook salmon adults that were classified as unsuccessful spawners and fell into the pre-spawning or premature spawning mortality categories, 2010–2015. The numbers above the bar are the sample sizes of fish classified as unsuccessful spawners.

Figure 5.5. Number and percentage of fall Chinook adults classified as pre-spawning mortality or premature spawning (empty redd producers). Based on current temperature exposures (Source: Connor et al. 2018).

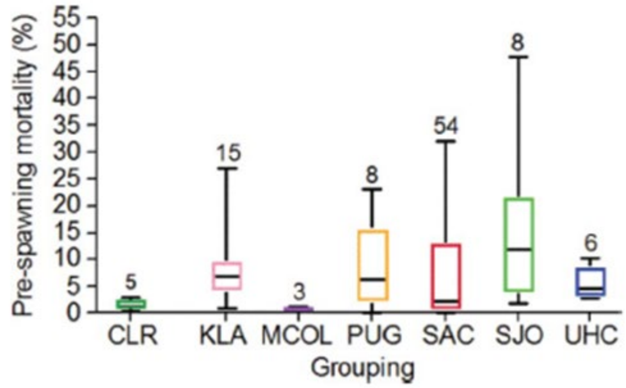


Figure 28. Pre-spawning mortality (%) distributions (5th, 25th, 50th, 75th, and 95th percentiles) for basin-level groupings of fall Chinook salmon calculated with empirical data from Bowerman et al. (2016) compared to the distributions calculated from simulations made with the somatic energy use model for the PIT-tagged, hatchery-origin fall Chinook salmon adults in the Upper Hells Canyon spawning area (UHC). The numbers above the 95th percentile are the sample sizes. Other abbreviations: CLR, Clearwater River lower reach, ID; KAL, Klamath Falls, OR; MCOL, Mid-Columbia, OR; PUG, Puget Sound, WA; SAC, Sacramento, CA; SJO, San Joaquin, CA. The data were provided by the corresponding author of Bowerman et al. (2016), except for the Clearwater River lower reach that were taken from Arnsberg et al. (2007, 2010) and Arnsberg and Kellar (2007a, 2007b, 2007c).

Figure 5.6. Pre-spawning mortality estimates across the Pacific Northwest (Source: Connor et al. 2018).

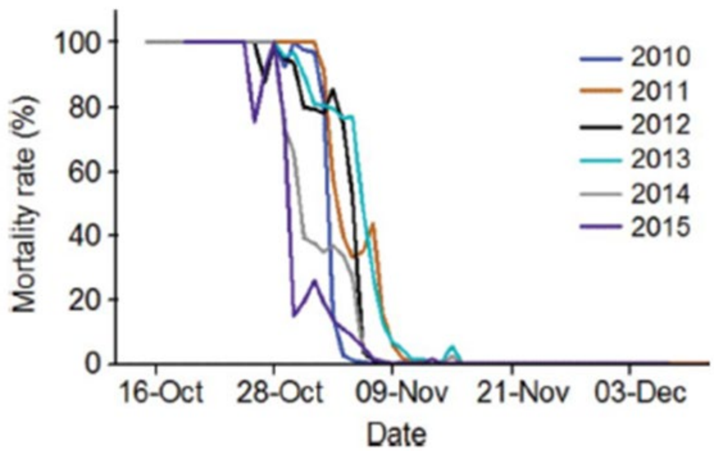


Figure 30. Simulated daily pre-spawning mortality rates (%) of PIT-tagged, hatchery-origin fall Chinook salmon adults along the Upper Hells Canyon spawning area, 2010–2015.

Figure 5.7. Pre-spawning mortality rates estimated for the Upper Hells Canyon reach above the Salmon River (Source: Connor et al. 2018).

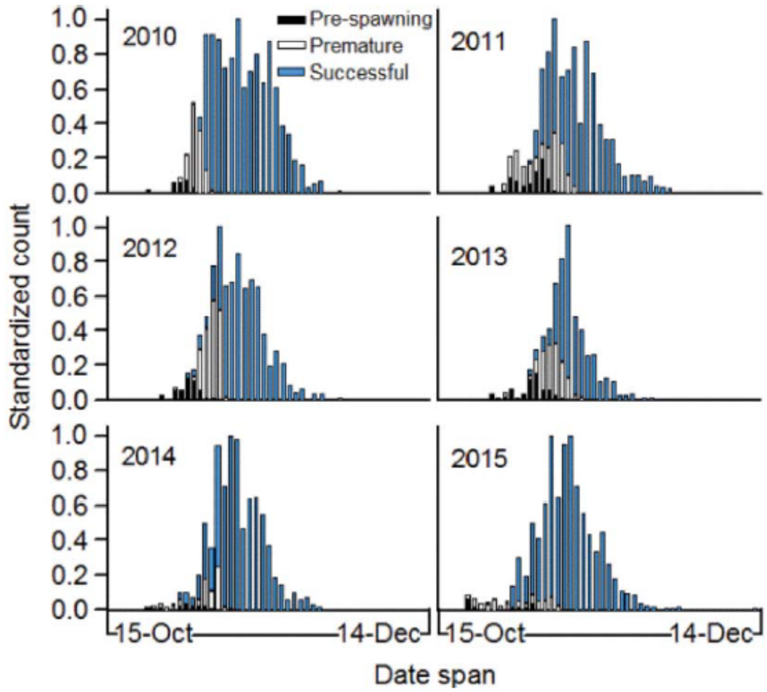


Figure 32. Standardized, simulated spawning initiation date distributions for PIT-tagged, hatchery-origin fall Chinook salmon adults that fell into the pre-spawning mortality category (had they survived to initiate spawning), into the category of premature spawning mortalities, or were classified as successful spawners, 2010–2015.

Figure 5.8. Estimated unsuccessful and successful spawning proportions for the Snake River (Source: Connor et al. 2018).

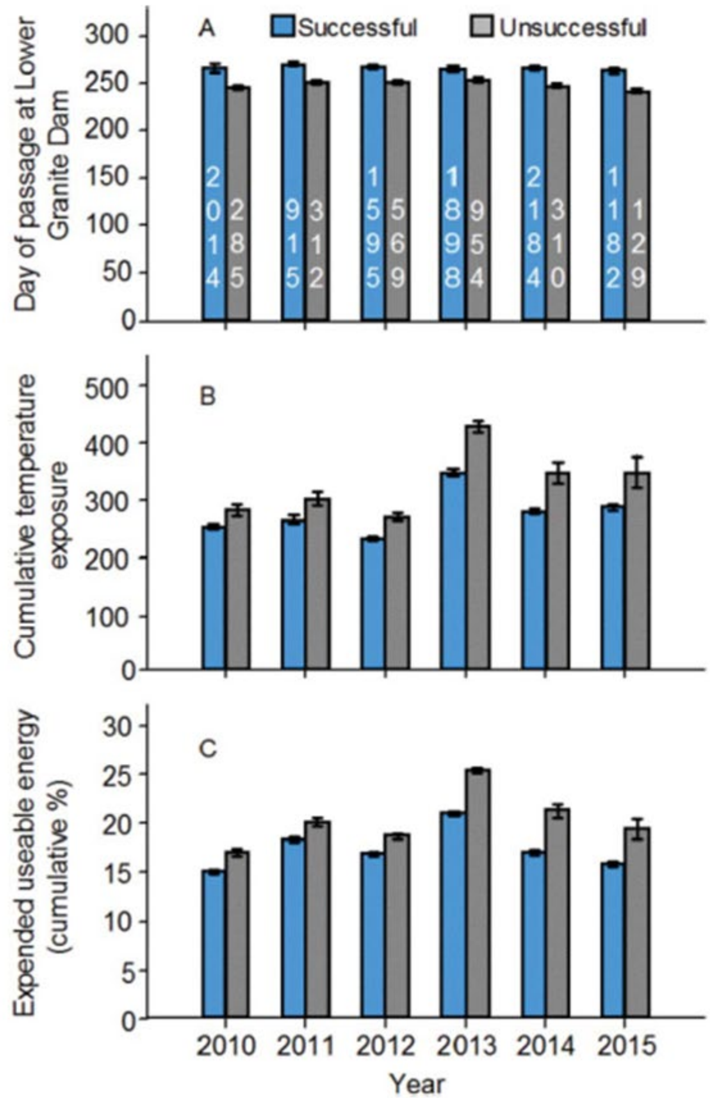


Figure 34. Annual mean (95% C.I) day of passage at Lower Granite Dam (panel a; numbers in bars = *n*), simulated cumulative temperature exposures between Bonneville and Lower Granite dams (panel b), and simulated expended useable energy (cumulative %; panel c) during Stages I and II for PIT-tagged, hatchery-origin fall Chinook salmon adults that were classified as either successful or unsuccessful spawners, 2010–2015.

Figure 5.9. Comparative estimates of unsuccessful and successful spawners passing from lower Granite (Source: Connor et al. 2018).

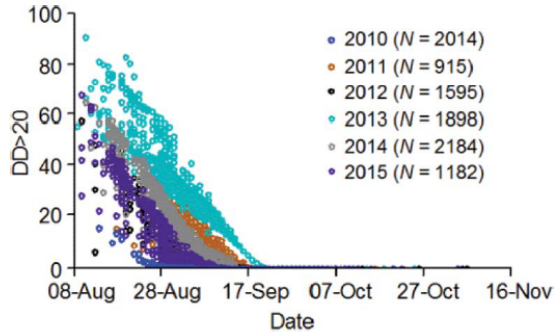


Figure 38. System-wide values of degree days above 20°C (DD>20) for PIT-tagged, hatchery-origin fall Chinook salmon adults that were classified as successful spawners by the somatic energy use model plotted against the observed dates of passage at Bonneville Dam of those adults, 2010–2015.

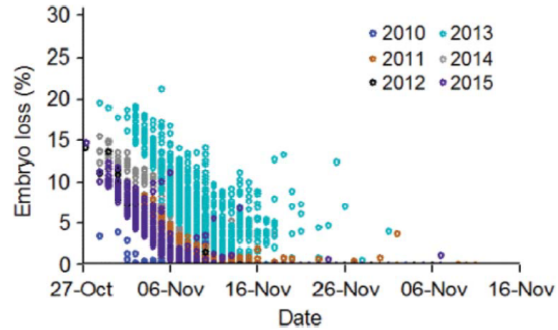


Figure 40. Simulated embryo loss (%) due to exposure to degree days above 20°C for PIT-tagged, hatchery-origin fall Chinook salmon adults that were classified as successful spawners by the somatic energy use model plotted against the simulated spawning initiation dates of those adults along the Upper Hells Canyon spawning area, 2010–2015.

Figure 5.10. Estimated temperature exposures for different yearly runs of successful spawners and estimated embryo losses due to exposures to prior degree days above 20°C for those successful spawners for the Upper Hells Canyon spawning area (Source: Connor et al. 2018).

Likewise, Connor et al. 2018 identified the Hells Canyon Reach as the second hottest spawning reach for fall Chinook, and stated, "Daily mean temperature recorded after 01-Sep along the Lower Hells Canyon spawning area exceeded 20°C 1 time in 2010, 27 times in 2011, 14 times, in 2012, 23 times in 2013, 17 times in 2014, and 11 times in 2015. Fish entering secondary or tertiary spawning areas can sometimes be exposed to temperatures above 20°C after tributary entry within the Lower Snake River watershed (Figure 3 of Connor et al. 2018). For example, in 2013, daily mean temperature exceeded 20°C in the Grande Ronde, Imnaha, and Salmon lower reaches 17, 16, and 18 times, respectively." Fall chinook adults must migrate over exceedingly long distances, with over 500 km traversed by the time they cross Lower Granite Dam (Connor et al. 2018).

According to Jensen et al. 2006, gametes held during suboptimal migration temperatures (from Summer Chinook adults held in temperatures comparable to those found during migration in this section of the Snake River prior to the time of application of the criterion) resulted in a higher rate of egg mortality. Jensen et. al. 2006 held adult Summer Chinook salmon at 16-19°C temperature for 21 days two weeks prior to spawning, which resulted in a 10.3% and 8.7% increase in egg mortality compared to salmon held at cooler temperatures. In addition, the study estimated significant pre-spawning excess mortality of 49%. Although such quantitative information is not available for the Snake River Fall Chinook population, such studies provide evidence that reduced gamete quality due to suboptimal migration temperatures can compromise eggs to the point of significant loss. Cold water refuge access is exceedingly important to Fall Chinook during migration (Keefer et al. 2018, Chapter 4), and therefore, minimization of such impacts to gametes through tools such as preservation of refuges is of paramount importance given the significant impact such migration and holding temperatures can have on productivity. There is evidence that a number of potential refuges is located in this stretch of the Snake River, as mentioned in the Recovery Plan (NOAA 2017) and the 401 certification application (Idaho

Power, 2018) (see Figure 4.4). However, a review of the refuges, quality, accessibility and sufficiency to mitigate effects on migrating adults and gametes has not been conducted to date.

For Critical Habitat, the PBFs most relevant to the Agency's action are water temperatures sufficient for fall-run Chinook adult migration, adult spawning, embryo incubation, and alevin development (see Table 3.3), which, as described above, may be likely adversely affected by this action.

Based upon the USEPA's review, the Agency's Action is **likely to adversely affect** Snake River fall Chinook and its Critical Habitat.

5.4.5. <u>Southern Resident Killer Whale</u>

5.4.5.1. Presence in the Hells Canyon Reach of the Snake River

As established in Chapter 3, Southern Resident Killer Whale ESU predates upon threatened and endangered species present in the action area.

5.4.5.2. Timing in the Hells Canyon Reach of the Snake River

As established in Chapter 3, Southern Resident Killer Whale ESU predates upon threatened and endangered species present in the action area during the time of the SSC application.

5.4.5.3. Direct and indirect effects

Indirect potential adverse effects on SRKW may occur from the Agency's Action. As discussed in Chapter 3, SR fall Chinook is one of the top three major prey species for SRKW. Further, as described above, SR fall Chinook is likely to be directly adversely affected by the Agency's Action. The USEPA does not have data and information available to accurately project to what extent adult fall Chinook production and interception by SRKW in the ocean could be affected by this action using a population model or other quantification method. However, the Agency has quantified the potential effects of the Agency's Action for the freshwater migratory adult and egg and fry lifestages of SR fall Chinook and found that the Agency's Action will likely result in adverse effects on SR fall Chinook.

The Agency is not able to quantify with certainty the impacts to a major SRKW prey, SR-produced fall Chinook adults present in the ocean, based upon the estimated adverse impacts to fall Chinook in the freshwater environment. Due to this uncertainty and the likely adverse effects on the freshwater lifestages of SR fall Chinook identified above, the USEPA is applying a conservative precautionary principle and concluding that the Agency's action **is likely to adversely affect** Southern Resident Killer Whales. During pre-consultation discussions, NOAA has indicated to the USEPA that the prey abundance for SRKW may not be significantly diminished as a result of the Agency's Action based on the recent status assessment (NOAA, 2016). Absent an analysis of smolt production, which may be provided in NOAA's Biological Opinion on this consultation, USEPA does not have the scientific bases to make a conclusion other than likely to adversely affect Southern Resident Killer Whales.

5.4.6. SR Bull Trout

5.4.6.1. Presence in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Bull Trout and its critical habitat are present in the Snake River.

5.4.6.2. Timing in the Hells Canyon Reach of the Snake River

As established in Chapters 3 and 4, Bull Trout are present in the Snake River at this time of year, with the range in outmigration dates for Bull Trout leaving the Imnaha River to forage overlapping with the time of the SSC change (Table 3.6).

5.4.6.3. Direct and indirect effects

As described in Chapter 3, adult and subadult bull trout require cold water, with thermal tolerances lower than other adult salmonids. Adult and subadult bull trout use this portion of the Snake River as foraging, migration, and overwintering FMO habitat. As described in Chapters 3 and 4, the largest population of adult bull trout in the Action Area migrate downstream from the Imnaha River to the mainstem Snake River to feed in the fall to winter months (Figure 3.9). Although typical outmigration patterns from the Imnaha represent a broad distribution, with a few individuals leaving the Imnaha for the Snake River in late September through the end of the October, the peak of the distribution is skewed to later November, with mean dates ranging from mid to late November for the pit-tag tracking data available, 2011-2014. The last receiver for the pit-tag data is at Rkm 7 of the Imnaha River. During the start of outmigration, Imnaha River temperatures are below 14C, and below 10°C (both as 7dadm) at the peak time (Figure 4.6). The 75th percentile 7dadm temperature for the Imnaha River from October 23rd to November 6th is 9.6C, whereas the section of the Snake River that the Imnaha enters ranges from 17.5-19.5°C as the 75th percentile of the range of data collection (range depends on station; Figure 4.4). The USEPA recommended criterion in its Temperature Guidance (USEPA 2003) to protect bull trout adult and subadult foraging is 16°C with sufficient access to cold water refuges. Likewise, in its 2015 Opinion on USEPA's Action on the application of a 16°C criterion to bull trout adult foraging, migration and overwintering (FMO) habitat, Fish and Wildlife Service found that 16°C was not likely to adversely affect bull trout or FMO habitat (USFWS 2015). Although meeting 14.5°C as a 7dadm during the time of application would mean improvement to Snake River temperatures which typically currently exceed 14.5°C as a 7dadm in the Snake River at and above where the Imnaha enters at this time of year, the change in the criterion may affect bull trout by delaying outmigration by up to one week compared to outmigration should waters meet the 13°C criterion, and access to cold water refuges therefore must be adequate to minimize such effects. However, based upon the available data, the Imnaha River can serve as an adequate cold water refuge during the early outmigration period. Therefore, any effects are likely to be insignificant. Likewise, the PBFs for critical habitat that are most relevant to this action are #2 and #5 (Section 3.8.5).

2. Migration habitats with minimal physical, biological, or water quality impediments between spawning, rearing, overwintering, and freshwater and marine foraging habitats, including but not limited to permanent, partial, intermittent, or seasonal barriers; and

5.Water temperatures ranging from 2 to 15°C (36 to 59°F), with adequate thermal refugia available for temperatures that exceed the upper end of this range.

For #2 and #5, this action is unlikely to result in significant effects to bull trout critical habitat. Although some effects are expected, we do not expect these to be significant for the migration foraging habitat at the time it is in use, and for 2) we have identified sufficient cold water refuge availability for Imnaha bull trout to minimize the effect of temperatures in excess of 15℃.

Based upon this review, the USEPA finds that its action **may affect but is not likely to adversely affect** SR Bull Trout and its critical habitat.

6. CUMULATIVE EFFECTS AND ONGOING ENVIRONMENTAL EFFECTS, AND UNCERTAINTY

6.1. Cumulative and Ongoing Environmental Effects

6.1.1. Dissolved Oxygen

As described in Chapter 4, "Baseline," dissolved oxygen levels in the reach are frequently below the ambient water quality criteria that the states of Idaho and Oregon have put in place to protect salmonid spawning. For Oregon, there is an 11 mg/L requirement (with 9 mg/L as a minimum allowed if intergravel dissolved oxygen is at least 8 mg/L) established to protect salmonid spawning; whereas Idaho has established a minimum of 6.5 mg/L as a SSC for dissolved oxygen below the Hells Canyon Dam (Figure 4.7 and Figure 4.8). As described in Geist et al. 2006, inadequate levels of dissolved oxygen, such as those found in the Snake River in October and early November, can result in reduced thermal tolerances for eggs and fry.

6.1.2. Gravel Quality

As summarized in Connor et al. 2018, gravel quality has a significant impact on the ability of eggs to incubate properly and mature to produce well-developed fry (Figure 6.1and Figure 6.2). Likewise, Jensen et al. (2009) found similar relationships between eggs and fry development and survival linked to gravel quality[15]. Due to the poor quality of gravels in a large portion of the Upper and Lower Hells Canyon Reach, there are potential cumulative adverse impacts for eggs to fry development and survival in the Action Area.

[15] Jensen, David W. , Steel, E. Ashley , Fullerton, Aimee H. and Pess, George R.(2009) 'Impact of Fine Sediment on Egg-To-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies', Reviews in Fisheries Science, 17: 3, 348 — 359, First published on: 01 January 2009

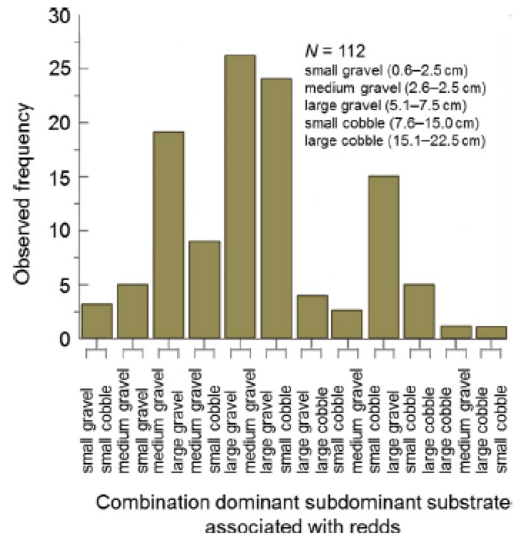


Figure 20. Frequency distribution of combined dominant (left) and subdominant (right) substrate types observed at fall Chinook salmon redds in the Upper and Lower Hells Canyon spawning areas, 1993–1995. The figure originally published in Groves and Chandler (1999) was provided for use by the editorial staff of Transactions of the American Fisheries Society. The font was modified and color added.

Figure 6.1. Relationship between gravel quality and fall Chinook redd distribution in the Hells Canyon Reach of the Snake River (Source: Connor et al. 2018)

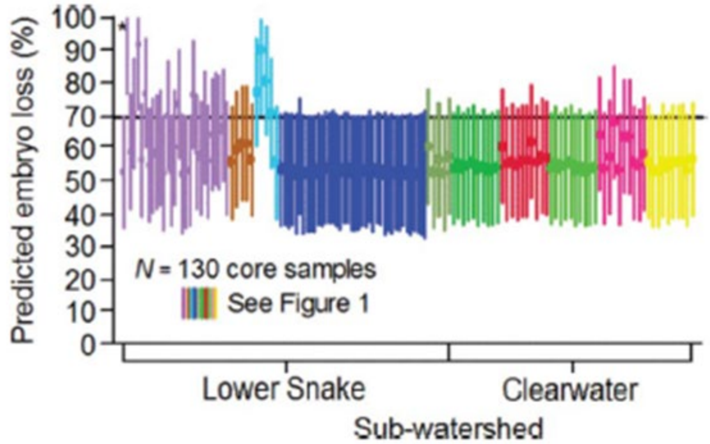


Figure 25. Embryo loss (%) predicted (± 95% CI) for the primary, secondary, and tertiary fall Chinook salmon spawning areas in the sub-watersheds that make up the upper stretch of the river system. The equation was: embryo loss (%) = -52.689 − $0.160^{*}S_{0.85} + 0.186^{*}S_{0.85}S_{0.85}$ ($N = 8$; $r^2 = 0.89$; $P = 0.004$); S0.85 was the mean percentage of fines less than 0.85 mm in diameter in each of the substrate samples collected by either Arnsberg et al. (1992) or Bennett et al. (2003). The prediction marked by an asterisk in panel was 105.1 ± 27.8%.

Figure 6.2. Embryo loss predicted for the primary (and secondary and tertiary) spawning areas of the Snake River (purple represents Upper Hells Canyon Reach) (Source: Connor et al. 2018)

6.1.3. <u>Climate Change</u>

Climate change has been and will continue to impact the Action Area and ESUs in a variety of ways. The impact of increasing criterion above thermal optima for certain lifestages, is to narrow the distribution of salmonid phenotypes to a narrower range. Decreasing diversity can impact the ability of salmonids to acclimate quickly and survive times of rapid change from year to year and season (e.g., Isaak et al. 2018; Crozier et al. 2008).

6.1.3.1. Future air and river temperatures and flows and implications

As summarized in Connor et al. 2018 and Keefer et al. 2018, the Snake River basin is projected to undergo major changes in air temperature, and likewise, Snake River and tributary temperatures are projected to increase, and flow magnitude and peak flow timing are also projected to change. From the data available for just below the Hells Canyon Outflow (Figure 5.1), 2010-2018 river temperatures are universally higher than those measured 1990-1999 (0.4-1.5C), which comports well with current trends estimated across large river systems in the Pacific Northwest (Chapter 4, Isaak et al. 2018), a 0.17°C increase per decade in larger river summer temperatures from the 1970s to present and 0.2°C per decade for the Columbia River basin. Future projections to 2080 for summer river temperatures are 1.7-2°C (Isaak et al. 2018 and Yearsley 2009) for the Columbia River and up 1-5°C across the Pacific Northwest (see literature compilation), which is comparable to the river temperature future simulations for the Upper Hells Canyon summarized in Connor et al. 2018 (Figure 6.3). Further Connor et al project that such high temperatures in the summer to fall timeframe will significantly impact every lifestage of fall Chinook present in the Snake River, from migration, to successful spawning, to fry emergence.

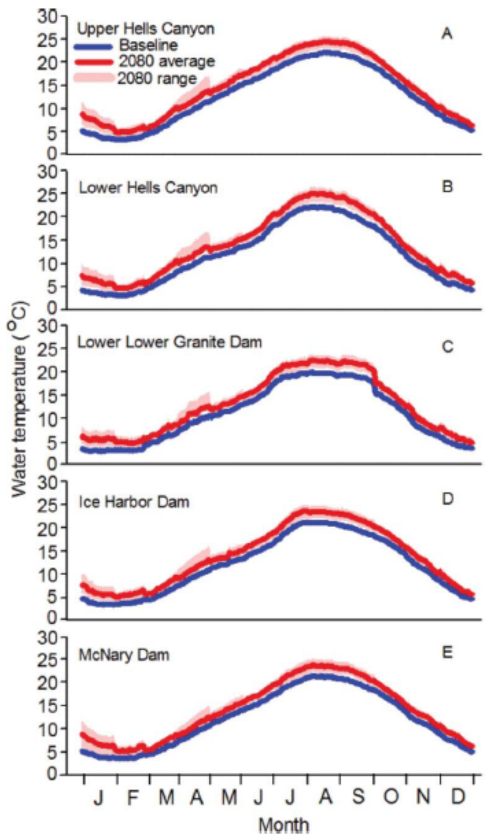


Figure 43. Baseline (1991–2006 average) and adjusted 2080 daily mean temperatures for Upper Hells Canyon (panel A), Lower Hells Canyon (panel B), Lower Granite Dam (panel C), Ice Harbor Dam (panel D), and McNary Dam (panel E). Refer to the Appendix to see how temperatures were simulated starting with information provided by Hamlet et al. (2010).

Figure 6.3. Baseline and 2080 predicted river temperatures in the Snake River Basin. Panels A and B include Upper and Lower Hells Canyon Reach temperatures. Dashed line indicates November 1 (Source: Connor et al. 2018).

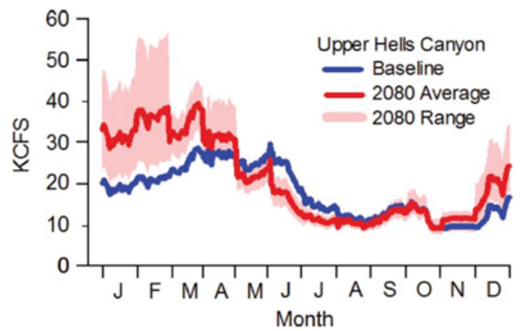


Figure 41. Baseline (1991–2006 average) and adjusted 2080 flows for Upper Hells Canyon from Hells Canyon Dam to the Imnaha River mouth. Refer to the Appendix to see how flows were simulated starting with information provided by Hamlet et al. (2010). The figure style is also attributed to Hamlet et al. (2010).

Figure 6.4. Baseline and 2080 projected flow changes in the Upper Hells Canyon Reach. Dashed line indicates November 1. (Adapted from Connor et al. 2018).

Fall Chinook Migration and Spawning Success and Spawning Timing Implications

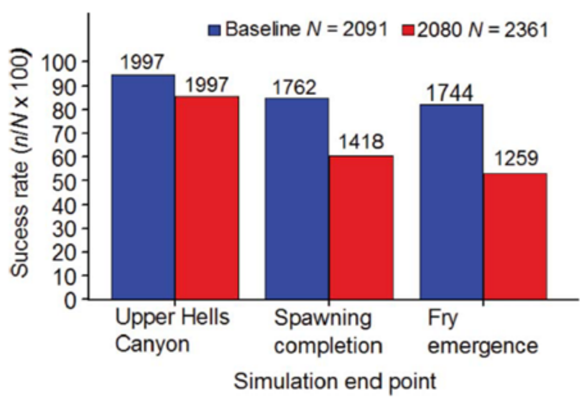


Figure 44. Simulated success rates (%) migration from Bonneville Dam to the Upper Hells Canyon spawning area, from Bonneville Dam to spawning completion, and from Bonneville Dam to fry emergence simulated for natural-origin fall Chinook salmon females under the baseline and 2080 scenarios, where *N* is the starting number of adults that arrived at Bonneville Dam and *n* is the number of adults that successfully completed each event given above the bars (except in the case of emergence; see text for description).

Figure 6.5. Simulations of 2080 conditions show reduced success rates for all aspects of migration, spawning completion and fry emergence lifestages for fall Chinook. (Source: Connor et al. 2018).

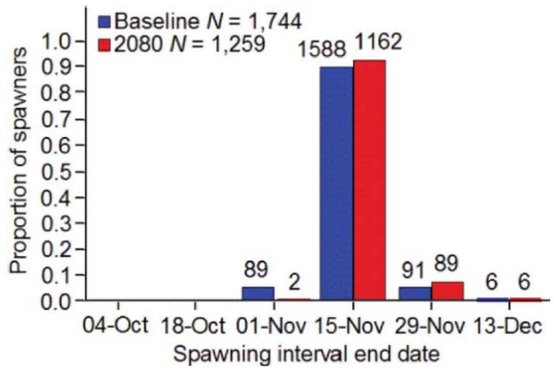


Figure 45. Effective spawn timing distributions simulated for natural-origin fall Chinook salmon females along the Upper Hells Canyon spawning area under the baseline and 2080 scenarios, where *N* is the total number of fish classified as successful spawners expressed in adult equivalents (see text), and *n* is the total number of adult equivalents in each spawning interval given above the bars.

Figure 6.6. Simulated 2080 fall Chinook spawning ability and distributions showing overall 28% reduction in successful spawners and shift in spawning intervals to after November 15 in comparison to current conditions when the majority occurs prior to November 15 (Source: Connor et al. 2018).

6.1.3.2. Ocean conditions, variability, and implications for species survival and health

Ocean conditions can significantly impact fall Chinook returning adult populations, in terms of total counts (survival through the FCRPS) and health of adults returning (Crozier et al. 2008). Starting in 2015, the effects of the "Blob," a significant heat event corresponding to reduced upwelling and poor forage for anadromous species while at sea has translated to poor counts for age 2 fall Chinook adults. Ocean conditions since 2015 have been poor, and indicators in 2019 are for much of the same conditions. Although in the past, ocean conditions cycled roughly with the Pacific Decadal Oscillation (Crozier et al. 2008), there are indications that such poor ocean conditions may become more prevalent in the future, but there is uncertainty about how frequently these events will occur and whether they will be intermittent or constant.

6.2. Downstream Protection

The currently applicable temperature criterion of 13°C that Idaho previously established to protect fall Chinook spawning only extends to the Salmon River. Likewise, this SSC action does not extend protection further downstream. However, Idaho waters extend below the Salmon River, and these waters include the designated use of salmonid spawning in Oregon. The State of Oregon has established criteria to protect salmonid spawning for this entire reach below the Hells Canyon Dam, downstream to the Washington and Idaho border. Although the USEPA's proposed approval of the 14.5°C criterion would result in a less stringent standard applicable below the Hells Canyon Dam to the Snake River, as described in Chapter 4, there is substantial redd emplacement in the Snake River below the Salmon River. These fall Chinook redds are not currently protected by spawning criteria in Idaho. However, because Oregon currently has applicable temperature criteria to protect salmonid spawning established for these waters, pursuant to 40 CFR Section 131.10(b), it is expected that Idaho will implement CWA programs to protect downstream uses, including for example, NPDES permits, TMDLs, and 401 certifications.

6.3. Other State Water Quality Standards that Influence the Effects of This Action

6.4. Hydrosystem Operations: Current and Future

The general effect of hydrosystem operations on stream temperature is that the large thermal mass created by the water stored in these reservoirs delays the peak summer water temperature to a later date and maintains temperatures at a higher level later into the fall relative to what would occur in a natural river condition. Reduced thermal complexity due to hydromodifications and reduced gravel quality also result from the hydrosystem emplacement. An additional stressor presented by the hydrosystem is continued prevention of wild fall Chinook, bull trout, and other species from migration to natal spawning habitat which is now blocked by the emplacement of the Hells Canyon Complex, which constrains spawning to a narrower area of suboptimal habitat below the Hells Canyon Dam. In the future, proposed work[16] that may be conducted in accordance with the draft 401 certification for the Hells Canyon Complex (once finalized) may provide for improved water quality and habitat upstream of the Complex, and water quality downstream, although those improvements will not be realized for several years.

6.5. Hatchery Operations

The threats from hatchery operations to Snake River fall Chinook are summarized as two primary issues: 1) the high proportion of hatchery fish as juveniles resulting limiting factors of competition with wild fish for habitat, food, and other resources, 2) high proportion of hatchery-origin spawners resulting in limiting factors of genetic change, loss of fitness, competition among spawners for resources, including spawning areas. Particularly factor 2 may cumulatively interact with the direct and indirect effects of this action by rendering fall Chinook wild spawning at a competitive disadvantage while undergoing additional stressors. In addition, the decrease in acclimation capacity that could result from this Action could render fall Chinook at a disadvantage in competing with hatchery Chinook.

6.6. Uncertainty

As mentioned in Chapter 5, there is a high degree of uncertainty due to the paucity of studies conducted on fall Chinook spawning and gamete effects in total, and in particular, in a declining (late fall) thermal regime. The potential adverse effects identified here for fall Chinook are based on five studies, with three providing some information on thresholds for eggs and fry mortality. However, none of the studies reflect the impacts to fall Chinook from conditions that would be allowed under the criterion for adults that migrate the length of the Columbia and Snake Rivers to spawn in the Upper Hells Canyon Reach. Connor et al. 2018 comes closest to capturing effects of the current thermal regime, but in its simulation, Connor et al. relies on egg mortality thresholds solely from Geist et al. 2006, which likely underestimates mortality in eggs and fry due to better than ambient holding temperatures in the Geist et al. experiments. Therefore, the USEPA considers the reported

[16] Idaho Power 2018

mortality threshold rates for fall Chinook eggs and fry to be likely underestimates and has therefore included upper bound estimates of potential egg mortality here.

Further, cumulative redd counts are based on when surveys are conducted and therefore, it is difficult to compare cumulative counts from one year to the next. For example, after a count on 10/23, more redds could be emplaced through10/29, however, they may not be counted until a later survey date, e.g early November. Likewise, redd counting methods employing drones began in recent years, and the results have been different than in previous years (Connor et al. 2017), which has made comparison of later redd counts with earlier redd counts difficult.

It is presumed that the Jensen et al. 2006 study on Summer Chinook translates to potential losses for fall Chinook. This is effect is underscored by the estimates in Connor et al. 2018. However, the estimates are based on a thermal regime in each study that is similar to but not equivalent to that which would be allowable due to the Agency's Action. Therefore, these estimates are uncertain.

For SRKW, there is much uncertainty surrounding the conversion of losses of eggs and fry estimated here, and conversion to SRKW prey (adult fall Chinook) associated with the Hells Canyon reach of the Snake River. Due to these uncertainties, the USEPA has not quantitatively translated the loss of eggs and fry to losses of adult fall Chinook prey for SRKW.

Lastly, particularly for Bull trout, Steelhead, and adult fall Chinook, refuge availability is of critical importance. Although refuges have been identified, their sufficiency has not been quantified, and further work to quantify, protect and restore adequate refuges is important for the future preservation of these species.

7. CONCLUSIONS/SUMMARY

Table 7.1. Summary of Effects Determinations for USEPA's Proposed Approval of Idaho's SSC for the Snake River Below Hells Canyon Dam to the Confluence with the Salmon River

Species	ESU	Status	Critical Habitat	Agency with Purview	USEPA Determination for Species	USEPA Determination for Critical Habitat
Bull trout (*Salvelinus confluentus*)	Columbia River DPS	Threatened 75 FR 63973 (10/18/10)	Designated	USFWS	NLAA	NLAA
Steelhead (*Oncorhynchus mykiss*)	Snake River Basin	Threatened 71 FR 834 (01/05/06)	Designated	NMFS	NLAA	NLAA
Chinook salmon (*O. tshawytscha*)	Snake River Spring/ Summer Chinook salmon	Threatened 70 FR 37160 (06/28/05)	Designated	NMFS	NLAA	NLAA
Chinook salmon (*O. tshawytscha*)	Snake River Fall-Run Chinook Salmon	Threatened 70 FR 37160 (06/28/05)	Designated	NMFS	LAA	LAA
Sockeye salmon (*O. nerka)*	Snake River Sockeye Salmon	Endangered 70 FR 37160 (06/28/05)	Designated	NMFS	NLAA	NLAA
Killer Whale (*Orcinus orca*)	Southern Resident DPS	Endangered 70 FR 69903 (11/18/05)	Designated*	NMFS	LAA	NLAA

**Designated critical habitat does not include the action area.*

8. ESSENTIAL FISH HABITAT

In this section, Essential Fish Habitat (EFH) is assessed for potential adverse impacts from the USEPA's proposed approval of the revised Idaho water quality standard for temperature for the Snake River below the Hells Canyon Dam to its confluence with the Salmon River.

8.1. Background

The Magnuson-Stevens Fishery Conservation and Management Act (MSA), as amended by the Sustainable Fisheries Act of 1996 (Public Law 104-267), requires federal agencies to consult with NOAA Fisheries on activities that may adversely affect EFH. According to the Magnuson-Stevens Fishery Conservation and Management Act (MSA§3), EFH means those waters and substrate necessary to fish for spawning, breeding, feeding, or growth and maturity. For the purpose of interpreting this definition of EFH: “waters” include aquatic areas and their associated physical, chemical, and biological properties that are used by fish; “substrate” includes sediment, hard bottom , structures underlying the waters, and associated biological communities; “necessary” means the habitat required to support a sustainable fishery and the managed species’ contribution to a healthy ecosystem; and “spawning, breeding, feeding, and growth to maturity” covers a species’ full life cycle (50 CFR 600.01). “Adverse effect” means any impact which reduces quality and/or quantity of EFH, and may include direct (e.g. physical disruption), indirect (e.g. loss of prey), site-specific or habitat-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810).

Pursuant to the MSA, the Pacific Fisheries Management Council (PFMC) has designated EFH for three species of federally-managed Pacific salmon: Chinook (*Oncorhynchus tshawytscha*); coho (O. *kisutch*); and Puget Sound pink salmon (O. *gorbuscha*) (PFMC 2000). Freshwater EFH for Pacific salmon includes all those streams, lakes, ponds, wetlands, and other water bodies currently, or historically accessible to salmon in Washington, Oregon, Idaho, and California, except areas upstream of certain impassable man-made barriers (as identified by PFMC 2000), and longstanding, naturally-impassable barriers (i.e. natural waterfalls in existence for several hundred years).

The objective of this EFH assessment is to determine if the proposed action may “adversely affect” designated EFH for relevant commercially or federally managed fisheries species within the proposed action area. It also describes conservation measures proposed to avoid, minimize or otherwise offset potential adverse effects to designated EFH resulting from the proposed action.

USEPA reviewed the NMFS information and (www.fisheries.noaa.gov/resource/map/essential-fish-habitat-mapper) to determine if the Action Area for this BE overlaps with EFH. In this case this overlap would be restricted to the EFH species that use freshwater habitats—Chinook, Pink salmon, and Coho salmon, since the this proposed modification to the water temperature in the Hells Canyon is not relevant to the Puget Sound (or other marine waters). The USEPA made the following conclusions:

--Pink Salmon. The Hells Canyon Reach of the Snake River is not within the distribution of O. *gorbusha*.

--Chinook salmon. The Hells Canyon Reach has been designated as EFH for Chinook. As described in Sections and   of the BE, Chinook salmon are present and use the Action Area as a migration corridor, spawning habitat, rearing, and smolt out-migrate. Other waters in the vicinity are also designated including, the Lower Snake-Asotin, Salmon River, and Clearwater River basins.

--Coho salmon. The Hells Canyon Reach of the Snake River is not within the distribution of coho salmon.

8.2. Description of the Project/Proposed Activity

The activity under consideration for this EFH assessment is identical to the description contained in the Biological Evaluation (BE) for this permit, located in Sections 1 and of this document.

Water quality is an important component of EFH. The potential effects of this action on EFH within the Action Area are the same as those described for fish species of concern in Sections 5 and . A summary of the determinations made for ESA listed species is found in Section . The USEPA has performed an assessment of how this action will affect the water temperature that Snake River Chinook could potentially be exposed to in the Hells Canyon Reach of the Snake River. A summary of the determinations made for ESA-listed species is found in the BE. Surface water criteria described in the permit provide restrictions to prevent harm to life stages of threatened and endangered species in the action area. However, using the information presented in the BE, the USEPA has determined that approval of this site-specific water quality criteria is likely to adversely affect Snake River Chinook salmon and their critical habitat. Therefore, this action is **likely to adversely affect Chinook salmon EFH** in this area.

8.3. EFH Conservation Measures and Conclusion

The USEPA concludes that the proposed action is likely to adversely affect EFH for Chinook salmon but will not affect EFH for coho or Puget Sound pink salmon.

9. LITERATURE CITED

Bain, D. 1990. Examining the validity of inferences drawn from photo-identification data, with special reference to studies of the killer whale (Orcinus orca) in British Columbia. Report of the International Whaling Commission, Special Issue 12:93-100.

Baird, R.W. 2000. The killer whale: foraging specializations and group hunting. P. 127-153 in J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead (editors). Cetacean societies: field studies of dolphins and whales. University of Chicago Press, Chicago.

Baird, R.W., Hanson, M.B., Dill, L.M. 2005. Factors influencing the diving behaviour of fish-eating killer whales: sex differences and diel and interannual variation in diving rates. Can J Zool 83, 257-267.

Barrett-Lennard LG, Heise KA. 2007. The natural history and ecology of killer whales. In: Estes J, ed, Whales, Whaling, and Ocean Ecosystems. pp 161-171.

Battin, J., M.W. Wiley, M.H. Ruckelshaus, R.N. Palmer, E. Korb, K.K. Bartz, and H. Imaki. 2007. Projected impacts of climate change on salmon habitat restoration. Proceedings of the National Academy of Sciences of the United States of America104(16):6720-6725. 6 pp.

Baxter, J. S. 1997. Aspects of the reproductive ecology of bull trout in the Chowade River, British Columbia. M.S. Thesis. University of British Columbia, Vancouver. 110 pp

Bigg, M. 1982. As assessment of killer whale (Orcinus orca) stocks off Vancouver Island, British Columbia. Report of the International Whaling Commission 32:655-666.

Birtwell, I.K. 1989. Comments on the sensitivity of salmonids to reduced levels of dissolved oxygen and to pulp mill pollution in Neroutsos Inlet, British Columbia. Can. Tech. Rep. Fish. Aquat. Sci. 1695. Fisheries and Oceans Canada, West Vancouver, BC

Bjornn et al. 1968. Migration and survival of Redfish Lake, Idaho, sockeye salmon, Oncorhynchus nerka. Trans. Am. Fish. Soc. 97:360-373.

Bjornn, T. and D. Reiser. 1991. Chapter 4: Habitat requirements of salmonids in streams. Pages 83-138 in W. Meehan, editors. Influences of Forest and Rangeland Management on Salmonids Fishes and Their Habitat. American Fisheries Society Special Publication 19.

Bond, C.E. 1992. Notes on the nomenclature and distribution of the bull trout and the effects of human activity on the species. Pages 1-4 in Howell, P.J. and D.V. Buchanan editors: Proceedings of the Gearhart Mtn. Bull Trout Workshop, Corvallis. Oregon Chapter of the American Fisheries Society. Corvallis, Oregon. 67 pp.

Bowerman T. et al. 2006. Pacific Salmon Prespawn Mortality: Patterns, Methods, and Study Design Considerations. Fisheries. Vol. 41(12). December 2016

Bonneville Power Administration (BPA). 1992. Stock summary reports for Columbia River anadromous salmonids, 5 volumes. Columbia River Coordinated Information System (CIS). (Available from Bonneville Power Administration, P.O. Box 3621, Portland, OR 97208-3621.)

Brett, J.R. 1964. The respiratory metabolism and swimming performance of young sockeye salmon. J. Fish. Res. Board Can. 21:1183-1226.

Brewin, P.A. and M. K. Brewin. 1997. Distribution Maps for Bull Trout in Alberta. Pages 206-216 in Mackay, W.C., M.K. Brewin and M. Monita, editors. Friends of the bull Trout Conference Proceedings. 10 pp.

Brown, L. G. 1992. Draft management guide for the bull trout on the Wenatchee National Forest. Prepared for Wenatchee National Forest by Washington Deptpartment of Wildlife, Wenatchee, Washington. 113 pp.

Buchanan, D., M. Hanson, and R.M. Hooten. 1997. Status of Oregon's Bull Trout, distribution, life history, limiting factors, management considerations and status. Technical Report, Report to Bonneville Power Administration, Contract No. 1994BI34342, Project No. 199505400, 185electronic pages (BPA Report DOE/BP-34342-5). 168 pp.

Burgner, R.L. 1991. Life history of sockeye salmon. Pages 1-118 in C. Groot and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver, BC.

Busby, P.J., T.C. Wainwright, G.J. Bryant, L. Leirheimer, R.S. Waples, F.W. Waknitz, and I.V. Lagomarsino. 1996. Status review of west coast steelhead from Washington, Idaho, Oregon, and California. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-NWFSC-27. 281 p.

Carlson, A. R. and L.J. Herman. 1974. Effects of lowered dissolved oxygen concentrations on channel catfish (Ictalurus punctatus) embryos and larvae. Trans. Amer. Fish. Soc. 103:623-626.

Carlson, A. R. and R.E. Siefert. 1974. Effects of reduced oxygen on the embryos and larvae of lake trout (Salvelinus namaycush) and largemouth bass (Micropterus salmoides). J. Fish Res. Board Can. 31:1393-1396.

Carlson, A. R., J. Blocher, and L. J. Herman. 1980. Growth and survival of channel catfish and yellow perch exposed to lowered constant and diurnally fluctuating dissolved oxygen concentration. Progr. Fish Cult. 42:73-78. Center for Whale Research. 2018. Friday Harbor, WA. Available at: https://www.whaleresearch.com/orca-population

Cavender, T. M. 1978. Taxonomy and distribution of the bull trout, Salvelinus confluentus (Suckley), from the American Northwest. California Fish and Game 64:139-174.

CBFWA 1990. https://www.nwfsc.noaa.gov/publications/scipubs/techmemos/tm200/sum.htm

Chandler, J. A., R. A. Wilkison, and T. J. Richter. 2003. Distribution, status, life history, and limiting factors of redband trout and bull trout associated with the Hells Canyon Complex. In: Technical appendices for new license application: Hells Canyon Hydroelectric Complex. Boise, ID: Idaho Power Company. Technical Report E.3.1 7; Chapter 4. 168 p.

Chapman, D. W., A. Giorgi, M. Hill, A. Maule, S. McCutcheon, and six others. 1991. Status of Snake River chinook salmon. Prepared for Pacific Northwest Utilities Conference Committee by Don Chapman Consultants, Inc., Boise, ID.

Clabough, T. S. et al. 2018. Thermal exposure of adult Chinook salmon and steelhead: Diverse behavioral strategies in a large and warming river system. PLOS(one): Vol.13(9). SEP 21 2018.

Columbia River DART (Data Access in Real Time). 1996—. Online interactive database of the Columbia Basin Research group, University of Washington School of Aquatic & Fishery Sciences. Available at www.cbr.washington.edu (September 2008).

Cooney, T. and M. Ford. 2007. Snake River Fall Chinook. Communication from C. Toole (NMFS) from T. Cooney and M. Ford (NWFSC) RE: Some notes on Snake Fall Chinook and hatchery questions, 8/29/2007.

Connor, W.P., H.L. Burge, R. Waitt, and T.C. Bjornn. 2002. Juvenile life history of wild fall Chinook salmon in the Snake and Clearwater rivers. North American Journal of Fisheries 22:703-712.

Connor, W. P., and H. L. Burge. 2003. Growth of wild subyearling Chinook salmon in the Snake River. North American Journal of Fisheries Management 23:594-599.

Connor, W. P. and H. L. Burge. 2003. Growth of wild subyearling Chinook salmon in the Snake River. North American Journal of Fisheries Management 23:594-599.

Connor, W. P., J. G. Sneva, K. F. Tiffan, R. K. Steinhorst, and D. Ross. 2005. Two alternative juvenile life histories for fall Chinook salmon in the Snake River basin. Transactions of the American Fisheries 134:291-304.

Connor et al. 2017. Research, Monitoring, and Evaluation of Emerging Issues and Measures to Recover the Snake River Fall Chinook Salmon ESU BPA Project Number 199102900. Report created March 2017.

Connor, W.P., K.F. Tiffan, J.A. Chandler, D.W. Rondorff, B.D. Arnsberg and K.C. Anderson. 2018. Upstream migration and spawning success of chinook salmon in a highly developed, seasonally warm river system. Reviews in Fisheries Science & Aquaculture. ISSN:2330-8257.

Coutant, C. C., and R. R. Whitney. 2006. Hydroelectric system development: effects on juvenile and adult migration. Pages 249–324 in R. N. Williams, editor. Return to the river: restoring salmon to the Columbia River. Elsevier, Amsterdam.

Crozier, L. G., A. P. Hendry, P. W. Lawson, T. P. Quinn, et al. 2008. Evolutionary responses to climate change for organisms with complex life histories: evolution and plasticity in Pacific salmon. Evolutionary Applications 1(1):252-270.

Crozier, L.G., B.J. Burke, B.P. Sandford, G.A. Axel, and B.L. Sanderson. 2014. Adult Snake River sockeye salmon passage and survival within and upstream of the FCRPS. Report of the National Marine Fisheries Service to the U.S. Army Corps of Engineers. Portland, Oregon.

Center for Whale Research (CWR). 2016. Narrative Contract Report for Item T0002 SRKW Annual Census as of July 1, 2016.

Dauble, D. D., R. L. Johnson, and A. Garcia. 1999. Fall chinook spawning in tailraces of lower Snake River hydroelectric projects. Transactions of the American Fisheries Society 128:672-697.

Dauble, D. D. and D. R. Geist. 2000. Comparison of mainstem habitats for two populations of fall Chinook salmon in the Columbia River basin. Regulated Rivers: Research and Management 16:345–361.

Davis, G.E., J. Foster, C. E. Warren and P. Doudoroff .1963. The Influence of Oxygen Concentration on the Swimming Performance of Juvenile Pacific Salmon at Various Temperatures, Transactions of the American Fisheries Society, 92:2, 111-124.

Davis, J.C. 1975. Minimal dissolved oxygen requirements of aquatic life with emphasis on Canadian species: a review. J. Fish. Res. Bd. Can. 32: 2295-2332.

Donald, D.B. and D.J. Alger. 1993. Geographic distribution, species displacement, and niche overlap for lake trout and bull trout in mountain lakes. Canadian Journal of Zoology 71:238-247.

Ebersole, J.L., P.J. Wigington, Jr, S.G. Leibowitz, R.L. Comeleo, and J. VanSickle. 2015. Predicting the occurrence of cold-water patches at intermittent and ephemeral tributary confluences with warm rivers. Freshwater Science 34:111-124.

Erickson, A.W. 1978. Population studies of killer whales (Orcinus orca) in the Pacific Northwest: a radio-marking and tracking study of killer whales. U.S. Marine Mammal Commission, Washington, D.C.

Everest, F.H., and D.W. Chapman.1972. Habitat selection and spatial interaction by juvenile chinook salmon and steelhead trout in two Idaho streams. J. Fish. Res. Board. Can. 22:1035-1081.

Fisher, T. 2004. Recent estimates of ESU abundance. Memorandum from T. Fisher, Fisher Fisheries, Ltd. to J. Geiselman, BPA, Portland, OR. October 25.

Fisher, T. and R. Hinrichsen. 2004. Preliminary Abundance-Based Trend Results for Columbia Basin Salmon and Steelhead ESUs. Bonneville Power Administration, Division of Environment, Fish and Wildlife, Portland, OR. 33 pp.

Ford, J.K.B., G.M. Ellis, L.G. Barrett-Lennard, A.B. Morton, R.S. Palm, and K.C. Balcomb. 1998. Dietary specialization in two sympatric populations of killer whales (Orcinus orca) in coastal British Columbia and adjacent waters. Can. J. Zool. 76:1456-1471.

Ford, J.K.B. and G.M. Ellis. 2006. Selective foraging by fish-eating killer whales Orcinus orca in British Columbia. Marine Ecology Progress Series 316:185-199.

Ford, M.J. (Ed.), T. Cooney, P. McElhany, N. Sands, L. Weitkamp, J. Hard, M. McClure, R. Kope, J. Myers, A. Albaugh, K. Barnas, D. Teel, P. Moran, and J. Cowen. 2011. Status Review Update for Pacific Salmon and Steelhead Listed Under the Endangered Species Act: Pacific Northwest. U.S. Department of Commerce. NOAA Technical Memorandum NMFS-NWFSC-113. November 2011.

Ford, J.K.B., G.M. Ellis, and K.C. Balcomb. 2000. Killer whales: the natural history and genealogy of Orcinus orca in British Columbia and Washington State, second edition. UBC Press, Vancouver, British Columbia.

Ford, J.K.B., B.M. Wright, G.M. Ellis, and J.R. Candy. 2010a. Chinook salmon predation by resident killer whales: seasonal and regional selectivity, stock identity of prey, and consumption rates. Department of Fisheries and Oceans, Canadian Science Advisory Secretariat Research Document 2009/101.

FPC (Fish Passage Center). 2003. Fish Passage Center annual report - 2002. Fish Passage Center, Columbia River Basin Fish and Wildlife Authority, Portland, OR.

Fraley, J.J., and B.B. Shepard. 1989. Life history, ecology and population status of migratory bull trout (Salvelinus confluentus) in the Flathead Lake and River System, Montana. Northwest Science 63(4):133-143.

Fullerton, A. H., C. E. Torgersen,J. J. Lawler, R. N. Faux, E. A Steel, T. J. Beechie, J. L. Ebersole, and S. G. Leibowitz. 2015. Rethinking the longitudinal stream temperature paradigm: region-wide comparison of thermal infrared imagery reveals unexpected complexity of river temperatures. Hydrological Processes 29:4719-4737.

Geist, D. R., C. S. Abernethy, K. D. Hand, V. I. Cullinan, J. A. Chandler, and P. A. Groves. 2006. Survival, Development, and Growth of Fall Chinook Salmon Embryos, Alevins, and Fry Exposed to Variable Thermal and Dissolved Oxygen Regimes. Transactions of the American Fisheries Society 135:1462-1477.

Geist, D.R., E.V. Arntzen, C.J. Murray, K.E. McGrath, Y.J. Bott, T.P. Hanrahan. 2008. Influence of river level on temperature and hydraulic gradients in chum and fall Chinook salmon spawning areas downstream of Bonneville Dam, Columbia River. North American Journal of Fisheries Management 28(1):30–41.

Goetz, F. 1989. Biology of the bull trout, Salvelinus confluentus, a literature review. Willamette National Forest. Eugene, Oregon. 60 pp.

Goniea, T.M., M.L. Keefer, T.C. Bjornn, C.A. Peery, D.H. Bennett, and L.C. Stuehrenberg. 2006. Behavioral thermoregulation and slowed migration by adult fall Chinook salmon in response to high Columbia River water temperatures. Transactions of the American Fisheries Society 135(2):408-419.

Good T.P., R.S. Waples, and P. Adams (editors). 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-66. 598 pp.

Gustafson RG, Wainwright TC, Winans GA, Waknitz FW, Parker LT, Waples RS. Status review of sockeye salmon from Washington and Oregon. NOAA Technical Memorandum NMFS-NWFSC-33: Seattle, Washington, USA: U.S. Department of Commerce; 1997. p. 282.

Hanson, M.B., R.W. Baird, J.K.B. Ford, J. Hempelmann-Halos, D.M. Van Doornik, J.R. Candy, C.K. Emmons, G.S. Schorr, B. Gisborne, K.L. Ayres, S.K. Wasser, K.C. Balcomb, K. Balcomb-Bartok, J.G. Sneva, and M.J. Ford. 2010c. Species and stock identification of prey consumed by endangered southern resident killer whales in their summer range. Endangered Species Research 11:69-82.

Hanson, B., C. Emmons, M. Sears, and K. Ayres. 2010a. Prey selection by southern resident killer whales in inland waters of Washington during the fall and early winter. Unpublished report. Draft. October 30.

Hanson, M.B., and C.K. Emmons. 2010. Annual residency patterns of southern resident killer whales in the inland waters of Washington and British Columbia. Revised draft. October 30.

Hanson, M.B., C.K. Emmons, E.J. Ward, J.A. Nystuen, and M.O. Lammers. 2013. Assessing the coastal occurrence of endangered killer whales using autonomous passive acoustic recorders. J. Acoust. Soc. Am. 134:3486-3495.

Hart, J.L. 1973. Pacific fishes of Canada. Fisheries Research Board of Canada, Bulletin No. 180. 740 p.

Hassemer, P. F. 1992. Run composition of 1991-92 run-year Snake River summer steelhead measured at Lower Granite Dam. Report to the National Oceanic and Atmospheric Administration, Award NA90AA-D-IJ718.

Hauser, D.D., W. Hauser, M.G. Logsdon, E.E. Holmes, G.R. VanBlaricom, and R.W. Osborne. 2007. Summer distribution patterns of southern resident killer whales Orcinus orca: core areas and spatial segregation of social groups. Marine Ecology Progress Series 351:301-310.

Healey, M. C. 1991. Life history of Chinook salmon (Oncorhynchus tshawytscha). Pages 311–394 in C. Groot and L. Margolis, editors. Pacific salmon life histories. University of British Columbia Press, Vancouver.

Hebdon, J.L., P. Kline, D. Taki, and T.A. Flagg. 2004. Evaluating reintroduction strategies for Redfish Lake sockeye salmon

Hegg, J. C., B. P. Kennedy, P. M. Chittaro, and R. W. Zabel. 2013. Spatial structuring of an evolving life history strategy under altered environmental conditions. Oecologia DOI 10.1007/s00442-012-2564-9.

Henjum, M.G., J.R. Karr, D.L. Bottom, D.A. Perry, J.C. Bednarz, S.G. Wright, S.A. Beckwitt, and E. Beckwitt. 1994. Interim protection for late-successional forests, fisheries, and watersheds. National forests east of the Cascade Crest, Oregon, and Washington. A report to the Congress and President of the United States Eastside Forests Scientific Society Panel. American Fisheries Society, American Ornithologists Union Incorporated, The Ecological Society of America, Society for Conservation Biology, The Wildlife Society. The Wildlife Society Technical Review 94-2. 112 pp.

Herger, L.G., G.A. Hayslip, and P.T. Leinenbach. 2007. Ecological Condition of Wadeable Streams of the Interior Columbia River Basin. EPA-910-R-07-005. U.S. Environmental Protection Agency, Region 10, Seattle, Washington.

Hickie, B.E., S, R.P., MacDonald, R.W., Ford, J.K.B. 2007. Killer whales (Orcinus orca) face protracted health risks associated with lifetime exposure to PCBs. Environ Sci Tech 41, 6613-6619.

Jeffries et al. 2012. Consequences of high temperatures and premature mortality on the transcriptome and blood physiology of wild adult sockeye salmon (Oncorhynchus nerka). Ecol Evol. 2012 Jul; 2(7): 1747–1764.

Jensen, J. O. T., W. E. McLean, W. Damon, and T. Sweeten. 2005. Puntledge River high temperature study: influence of high water temperatures on adult Chinook salmon (Oncorhynchus tshawytscha). Canadian Technical Report of Fisheries and Aquatic Sciences.

Jensen, J. O. T., W. E. McLean, T. Sweeten, W. Damon, and C. Berg. 2006. Puntledge River high temperature study: influence of high water temperatures on adult Chinook salmon

(Oncorhynchus tshawytscha) in 2004 and 2005. Canadian Technical Report of Fisheries and Aquatic Sciences 2662.

Jensen, David W., Steel, E. Ashley, Fullerton, Aimee H. and Pess, George R.(2009) 'Impact of Fine Sediment on Egg-To-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies', Reviews in Fisheries Science, 17: 3, 348 — 359, First published on: 01 January 2009

IDEQ. 2001. Middle Salmon River-Panther Creek subbasin assessment and TMDL. Idaho Department of Environmental Quality, Boise, ID.

IDEQ, and EPA. 2003. South Fork Clearwater River subbasin assessment and total maximum daily loads. Idaho Department of Environmental Quality and US Environmental Protection Agency.

IDEQ. 2011. Idaho's 2010 integrated report. Final. State of Idaho Department of Environmental Quality, Boise, ID.

IDEQ 2012- IDAPA 58-0102-1102 Docket located at: http://www.deq.idaho.gov/laws-rules-etc/deq-rulemakings/docket-no-58-0102-1102-final-rule/

IDEQ. 2014. Idaho's 2014 Integrated Report. Idaho Department of Environmental Quality. 625 pp. http://www.deq.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report.aspx [Accessed March 2017].

IDEQ. 2011. Idaho's 2010 integrated report. Final. State of Idaho Department of Environmental Quality, Boise, ID.

IDEQ. 2014. Idaho's 2014 Integrated Report. Idaho Department of Environmental Quality. 625 pp. http://www.deq.idaho.gov/water-quality/surface-water/monitoring-assessment/integrated-report.aspx [Accessed March 2017].

Idaho Power Company. 2018 (June). Clean Water Act Section 401 Water-quality certification application—Hells Canyon Complex, FERC Project No. 1971. Idaho Power, POB 70, Boise, Idaho.

Independent Scientific Group. 1996. Return of the river: restoration of salmonid fishes in the Columbia River ecosystem. Northwest Power Planning Council, Portland, Oregon. Publication 96-6. 584pp.

Irving, J.S., and T.C. Bjornn. 1981. Status of Snake River fall chinook salmon in relation to the Endangered Species Act. Idaho Cooperative Fishery Research Unit, University of Idaho, Moscow, for U.S. Fish and Wildlife Service.

Isaak, D.J., S. Wollrab, D. Horan and G. Chandler. 2012. Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. Climatic Change (2012) 113:499–524.

Isaak et al. 2018. Global Warming of Salmon and Trout Rivers in the Northwestern U.S.: Road to Ruin or Path Through Purgatory? Trans. Am. Fish. Volume147, Issue3 May 2018. Pages 566-587.

Keefer, M.L., C.A. Peery, and M.J. Heinrich. 2008. Temperature mediated en route migration mortality and travel rates of endangered Snake River sockeye salmon. Ecol. Freshwat. Fish 17(1):136-145.

Keefer ML, Clabough TS, Jepson MA, Johnson EL, Peery CA, Caudill CC (2018) Thermal exposure of adult Chinook salmon and steelhead: Diverse behavioral strategies in a large and warming river system. PLoS ONE 13(9): e0204274. https://doi.org/10.1371/journal.pone.0204274.

Krahn, M.M., Ford, M.J., Perrin, W.F., Wade, P.R., Angliss, R.P., Hanson, M.B., Taylor, B.L., Ylitalo, G.M., Dahlheim, M.E., Stein, J.E., Waples, R.S. 2004. 2004 status review of southern resident killer whales (Orcinus orca) under the Endangered Species Act. NOAA Technical Memorandum NMFS-NWFSC-62. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Seattle, WA.

Krahn, M.M., Hanson, M.B., Baird, R.W., Boyer, R.H., Burrows, D.G., Emmons, C.K., Ford, J.K.B., Jones, L.L., Noren, D.P., Ross, P.S., Schorr, G.S., Collier, T.K. 2007. Persistent organic pollutants and stable isotopes in biopsy samples (2004/2006) from southern resident killer whales. Mar Poll Bull 54, 1903-1911.

Krahn, M.M., Hanson, M.B., Schorr, G.S., Emmons, C.K., Burrows, D.G., Bolton, J.L., Baird, R.W., Ylitalo, G.M. 2009. Effects of age, sex and reproductive status on persistent organic pollutant concentrations in "southern resident" killer whales. Mar Poll Bull 58, 1522-1529.

Lachmuth, C.L., Alava, J.J., Hickie, B.E., Johannessen, S.C., Macdonald, R.W., Ford, J.K.B., Ellis, G.M., Gobas, F.A.C.P., Ross, P.S. 2010. Ocean disposal in resident killer whale (Orcinus orca) critical habitat: science in support of risk management. Fisheries and Oceans Canada.

Leary, R.F. and F.W. Allendorf. 1997. Genetic confirmation of sympatric bull trout and Dolly Varden in western Washington. Transactions of the American Fisheries Society 126:715-720.

Leathe, S.A. and P. Graham. 1982. Flathead Lake Fish Food Habits Study. Environmental Protection Agency, through Steering Committee for the Flathead River Basin Environmental Impact Study. 208 pp.

Mann, R. and C. Peery. 2005. Effects of Water Temperature Exposure on Spawning Success and Developing Gametes of Migrating Anadromous Fish. A report to Walla Walla District U. S. Army Corps of Engineers.

Mantua, N. J., S. R. Hare, Y. Zhang, J. M. Wallace, and R. C. Francis. 1997. A Pacific decadal climate oscillation with impacts on salmon. Bulletin of the American Meteorological Society 78:1069–1079.

Maret, T.R., T.A. Burton, G.W. Harvey, and W.H. Clark. 1993. Field testing of new monitoring protocols to assess brown trout spawning habitat in an Idaho stream. N. Am. J. Fish. Man. 13:567-580.

McClure, M., B. Sanderson, E. Holmes, C. Jordan, P. Kareiva, and P. Levin. 2000. Revised Appendix B of standardized quantitative analysis of the risks faced by salmonids in the Columbia River basin. National Marine Fisheries Service, Northwest Fisheries Science Center, Seattle, Washington.

McPhail, J.D., and J.S. Baxter. 1996. A Review of Bull Trout (Salvelinus confluentus) Life-history and Habitat Use in Relation to Compensation and Improvement Opportunities. University of British Columbia. Fisheries Management Report #104. 37 pp.

Muir, W.D., and J.G. Williams. 2012. Improving connectivity between freshwater and marine environments for salmon migrating through the lower Snake and Columbia River hydropower system. Ecological Engineering 48:19-24.

Myers, J. M., R.G. Kope, G.J. Bryant, D.J. Teel, L. J. Lierheimer, T. C. Wainwright, W. S. Grant, F. W. Waknitz, K. Neely, S. Lindley, and R. S. Waples. 1998. Status review of Chinook salmon from Washington, Idaho, Oregon, and California. U.S. Department of Commerce. Washington D.C.

National Academies of Sciences and Engineering. 1972. Water quality criteria 1972: report to the U.S. Environmental Protection Agency. 594 pp.

National Oceanic and Atmospheric Administration (NOAA). 1998. Endangered and Threatened Species: Proposed Endangered Status for Two Chinook Salmon ESUs and Proposed Threatened Status for Five Chinook Salmon ESUs; Proposed Redefinition, Threatened Status, and Revision of Critical Habitat for One Chinook Salmon ESU; Proposed Designation of Chinook Salmon Critical Habitat in California, Oregon, Washington, Idaho. Federal Register 63 (45): 11481-11520.

NMFS (National Marine Fisheries Service). 1995. Proposed recovery plan for Snake River salmon. NMFS, Protected Resources Division, Portland, Oregon.

National Oceanic and Atmospheric Administration (NOAA). 1992. Endangered and Threatened Species: Threatened Status for Snake River Spring/Summer Chinook Salmon, Threatened Status for Snake River Fall Chinook Salmon. Federal Register 57 (75):14653-14663.

NMFS (National Marine Fisheries Service). 2000. Biological Opinion. Reinitiation of Consultation on Operation of the Federal Columbia River Power System, Including the Juvenile

Fish Transportation Program, and 19 Bureau of Reclamation Projects in the Columbia Basin. National Marine Fisheries Service, Northwest Region, Seattle, WA. Endangered Species Act - Section 7 Consultation.

NMFS (National Marine Fisheries Service). 2004. Endangered Species Act – Section 7 Consultation Biological Opinion Consultation on Remand for Operation of the Columbia River Power System and 19 Bureau of Reclamation Projects in the Columbia Basin. NOAA's National Marine Fisheries Service (NOAA Fisheries) Northwest Region. Nov. 20, 2004.

NMFS. 2005 Updated Status of Federally Listed ESUs of West Coast Salmon and Steelhead. NOAA Technical Memorandum NMFS-NWFSC-66 U.S. National Oceanic and Atmospheric Administration National Marine Fisheries Service.

NMFS (National Marine Fisheries Service). 2008a. Recovery plan for southern resident killer whales (Orcinus orca). National Marine Fisheries Service, Northwest Regional Office. Seattle.

NMFS (National Marine Fisheries Service). 2008b. Supplemental comprehensive analysis of the Federal Columbia River Power System and mainstem effects of USBR Upper Snake and other tributary actions. National Marine Fisheries Service, Portland, Oregon.

NMFS (National Marine Fisheries Service). 2009a. Middle Columbia River steelhead distinct population segment ESA recovery plan. National Marine Fisheries Service, Northwest Region. Seattle.

NMFS (National Marine Fisheries Service). 2009b. Endangered Species Act – section 7 consultation biological opinion. Biological opinion on the effects of the Pacific coast salmon plan on the southern resident killer whale (Orcinus orca) distinct population segment. National Marine Fisheries Service, Northwest Region. Seattle.

NMFS (National Marine Fisheries Service). 2011b. Draft recovery plan for Idaho Snake River spring/summer Chinook and steelhead populations in the Snake River spring/summer Chinook salmon evolutionarily significant unit and Snake River steelhead distinct population segment. National Marine Fisheries Service. Portland, Oregon.

NMFS (National Marine Fisheries Service). 2011c. 2011 Report to Congress: Pacific Coastal Salmon Recovery Fund FY 2000 – 2010. National Marine Fisheries Service, Northwest Region. Portland, Oregon.

NMFS (National Marine Fisheries Service). 2011d. Southern resident killer whales (Orcinus orca) 5-year review: summary and evaluation. National Marine Fisheries Service, Northwest Region. Seattle.

NMFS (National Marine Fisheries Service). 2012d. Draft recovery plan for Oregon spring/summer Chinook salmon and steelhead populations. NMFS, West Coast Region. Portland, Oregon.

NMFS 2014. NMFS (National Marine Fisheries Service). 2014a. Supplemental recovery plan module for Snake River salmon and steelhead mainstem Columbia River hydropower projects. Portland, OR.

NMFS (National Marine Fisheries Service). 2015a. ESA recovery plan for Snake River sockeye salmon (*Oncorhynchus nerka*). National Marine Fisheries Service, West Coast Region. Portland, Oregon.

NOAA, NMFS. 2015. Proposed ESA Recovery Plan for Snake River Fall Chinook Salmon (*Oncorhynchus tshawytscha*). West Coast Region. October 2015.

NMFS. 2016. 2016 5-Year Review:Summary & Evaluation of Snake River Sockeye Snake River Spring-Summer Chinook Snake River Fall-Run Chinook Snake River Basin Steelhead. National Marine Fisheries Service Northwest Region. Portland, OR.

NMFS. 2016. 2016 5-Year Review: Summary & Evaluation of Middle Columbia River Steelhead National Marine Fisheries Service West Coast Region Portland, OR.

NOAA (National Oceanic and Atmospheric Administration). 2017. ESA recovery plan for Snake River fall Chinook salmon (*Oncorhymchus tshawytscha*). NOAA National Marine Fisheries Service. West Coast Region. Portland, OR. 366 pp.

NMFS. 2018a. FAQs SRKW & Columbia/Snake River Chinook Salmon Stocks. NOAA-Northwest Fisheries Science Center. 3 pp.

NMFS. 2018b. Southern Resident Killer Whales and West Coast Chinook Salmon. NOAA Fisheries West Coast Region - Northwest Fisheries Science Center. 8 pp.

NOAA Fisheries. 2017a. Killer whale (Orcinus orca) [online]. National Oceanic and Atmospheric Administration Fisheries. Updated January 2017. [Cited February 14, 2017.] Available from: http://www.nmfs.noaa.gov/pr/species/mammals/whales/killer-whale.html.

NOAA Fisheries. 2017b. Killer whales identification tips and vocalizations [online]. National Oceanic and Atmospheric Administration Fisheries. [Cited February 14, 2017.] Available from: http://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/killer_whale/id_vocalizations.html

NOAA 2019 – 2020 Adult Fall Chinook Forecast Available at: https://www.nwfsc.noaa.gov/research/divisions/fe/estuarine/oeip/g-forecast.cfm

NMFS. 2015. Endangered Species Act Biological Opinion on the EPA's Proposed Approval of Certain Oregon Water Quality Standards Including Temperature and Intergravel Dissolved Oxygen. National Marine Fisheries Service, Northwest Region, WCR: 2013-76, Portland, OR. 317 pp.

National Research Council. 1996. Dams and mitigation of their effects. Upstream. Chapter 9 *in* Upstream:Salmon and Society in the Pacific Northwest. NRC Committee on Protection and

Management of Pacific Northwest Anadromous Salmonids, Board on Environmental Studies and Toxicology. National Academy Press. Washington D.C.

Noren DP, Hauser DDW. 2016. Surface-based observations can be used to assess behavior and fine-scale habitat use by an endangered killer whale (Orcinus orca) population. Aquat Mamm 42(2):168-183.NWFSC. 2015. Status review update for Pacific salmon and steelhead listed under the Endangered Species Act: Pacific Northwest. Northwest Fisheries Science Center.

Olesiuk, P. F., M. A. Bigg, and G. M. Ellis. 1990. Life history and population dynamics of resident killer whales (Orcinus orca) in the coastal waters of British Columbia and Washington State. Reports of the International Whaling Commission 12:209-243.

Olesiuk, P. F., G. M. Ellis, and J. K. Ford. 2005. Life history and population dynamics of northern resident killer whales (Orcinus orca) in British Columbia. Department of Fisheries and Oceans Canadian Science Advisory Secretariat Research Document 2005/045.

Olson, P.A. and R. F. Foster (1957) Temperature Tolerance of Eggs and Young of Columbia River Chinook Salmon, Transactions of the American Fisheries Society, 85:1, 203-207.

Olson, P. A., R. E. Nakatani, and T. Meekin. 1970. Effects of thermal increments on eggs and young of Columbia River Fall Chinook. Pacific Northwest Laboratories, Richland, WA.

Osborne, R. W. 1999. A historical ecology of Salish Sea "resident" killer whales (Orcinus orca): with implications for management. Doctoral dissertation. University of Victoria, Victoria, British Columbia.

Peterson, W. T., J. L. Fisher, C.A. Morgan, S. M. Zeman, B. J. Burke, K. C. Jacobson. 2018. Ocean ecosystem indicators of salmon marine survival in the Northern California current. National Marine Fisheries Service, Northwest Fisheries Science Center. Newport, Oregon.

PFMC (Pacific Fisheries Management Council). 2000(FEB). Review of 1999 ocean salmon fisheries. Pacific Fishery Management Council. Portland, Oregon.

Phillips, R.W. and H.J. Campbell. 1962. The embryonic survival of coho salmon and steelhead trout as influenced by some environmental conditions in gravel beds. P. 60-73 in: 14th Annual Report. Pacific Marine Fisheries Commission. Portland, Oregon. 108 p.

Pratt, K.L. 1992. A Review of bull trout life history. 00. 5-9. In Proceedings of the Gearhart Mountain Bull Trout Workshop, ed. Howell, P.J. and D.V. Buchanan. Gearhart Mountain, Oregon. Corvallis, Oregon: Oregon Chapter of the American Fisheries Society. August 1992. 8 pp.

Quinn, T.P. 2005. The behavior and ecology of Pacific salmon and trout. American Fisheries Society and University of Washington, Seattle.

Randall, D.J. and J.C. Smith. 1967. the regulation of cardiac activity in fish in a hypoxic environment. Physiol. Zoo. 40:104-113.

Ratliff, D.E., and P.J. Howell. 1992. The status of bull trout populations in Oregon. Pages 10-17 in P.J. Howell and D.V. Buchanan, editors. Proceedings of the Gearhart Mountain bull trout workshop. Oregon Chapter of the American Fisheries Society, Corvallis. 80 pp.

Rhodes, J. J., D. A. McCullough, and J. F. A. Espinosa. 1994. A coarse screening process for evaluation of the effects of land management activities on salmon spawning and rearing habitat in ESA consultations. Columbia River Inter-Tribal Fish Commission. Technical Report, 94-1. Portland, Oregon.

Richter A., and S.A Kolmes. 2005. Maximum temperature limits for Chinook, coho, and chum salmon, and steelhead trout in the Pacific Northwest. Reviews in Fisheries Science 13:23-49.

Rieman, B.E., D.C. Lee and R.F. Thurow. 1997. Distribution, status and likely future trends of Bull trout within the Columbia River and Klamath River basins. North American Journal of Fisheries Management 17:1111-1125.

Rieman, B.E., and J.D. McIntyre. 1993. Demographic and habitat requirements of bull trout Salvelinus confluentus. General Technical Report INT-GTR- 302. U.S. Department of Agriculture, Forest Service, Intermountain Research Station, Ogden, Utah. 42 pp.

Rieman, B.E., D. Isaak, S. Adams, D. Horan, D. Nagel, C. Luce, D. Myers. 2007. Anticipated climate warming effects on bull trout habitats and populations across the Interior Columbia River Basin. Transactions of the American Fisheries Society 136:1552-1565.

Rombaugh, P.J. 1986. Mathematical model for predicting the DO requirements of steelhead (Salmo gairdneri) embryos and alevins in hatchery incubators. Aquaculture 59:119-137.

Rombough, P.J. 1988. growth, aerobic metabolism, and dissolved oxygen requirements of embryos and alevins of steelhead, Salmo gairdneri. Can. J. Zool. 66: 651-660, cited by Oregon Department of Environmental Quality (1995), Standards & Assessment Section, 1992-1994 Water Quality Standards Review, Dissolved Oxygen.

Schaller et al. 1999. Contrasting patterns of productivity and survival rates for stream-type chinook salmon (Oncorhynchus tshawytscha) populations of the Snake and Columbia rivers. Can. J. Fish. Aquat. Sci. 56: 1031–1045 (1999)

Scott, W. B., and E. J. Crossman. 1973. Freshwater fishes of Canada. Bulletin 184. Fisheries Research Board of Canada, Ottawa, ON, Canada.

Sedell, J.R., and J.L. Froggatt. 1984. Importance of streamside forests to large rivers: the isolation of the Willamette River, Oregon, USA from its floodplain by snagging and streamside forest removal. Internationale Vereinigung für Theoretische und angewandte Limnologie Verhandlungen 22:1828-1834.

Siefert, R.E. and W.A. Spoor. 1973. Effects of reduced oxygen on embryo s and larvae of the white sucker, coho salmon, brook trout and walleye. As cited in Welch, E.B. 1980. Ecological Effects of Waste Water. Cambridge University Press.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An ecosystem approach to salmonid conservation. ManTech Environmental Research Services, Inc. Corvallis, Oregon. National Marine Fisheries Service, Portland, Oregon.

Stoddard, J. L., D. V. Peck, et al. 2005. An ecological assessment of western streams and rivers. Prepared by Environmental Monitoring and Assessment Program (EMAP) team. EPA 600/R-05/005. U.S. Environmental Protection Agency. Washington, D.C.

Scheuerell, M. D., and J. G. Williams. 2005. Forecasting climate induced changes in the survival of Snake River spring/summer Chinook salmon (Oncorhynchus tshawytscha). Fisheries Oceanography 14(6):448–457.

Seymour, A. H. 1956. Effects of temperature upon young Chinook salmon. Ph. D. Thesis. University of Washington, Seattle. 127 pp.

Sowden, T.K. and G. Power. 1985. Prediction of rainbow trout embryo survival in relation to groundwater seepage and particle size of spawning substrates. Transactions of the American Fisheries Society 114:804-812.

Spence, B.C., G.A. Lomnicky, R.M. Hughes, and R.P. Novitzki. 1996. An ecosystem approach to salmonid conservation. ManTech Environmental Research Services, Inc. Corvallis,Oregon. National Marine Fisheries Service, Portland, Oregon.

Sprague, J.B. 1985. Factors that modify toxicity. in: fundamentals of aquatic toxicology, G.M. Rand and S.R. Petrocelli (eds.). Hemisphere Publishing, New York.

Tiffan, K. F. and W. P. Connor (2012). Seasonal use of shallow water habitat in the Lower Snake River reservoirs by juvenile fall Chinook salmon. 2010–2011 final report of research to the U.S. Army Corps of Engineers,Walla Walla District, Walla Walla, WA: 74.

Trites, A. W., and C. P. Donnelly. 2003. The decline of Steller sea lions Eumetopias jubatus in Alaska: a review of the nutritional stress hypothesis. Mammal Review 33(1):3-28.

USEPA. 1986b. Quality Criteria for Water [1986] (The Gold Book) EPA 440/5-86-001. Office of Water. Regulations and Standards.

USEPA. 1986a. Ambient Water Quality Criteria for Dissolved Oxygen - 1986. USEPA 440/5-86-003. USEPA Office of Water. Washington, DC.

USEPA. 2002. Issue Paper 5: Summary of Technical Literature Examining the Physiological Effects of Temperature Region 10 Temperature Water Quality CriteriaGuidance Development Project. USEPA Region 10 Office of Water, Seattle, WA.

USEPA. 2003. EPA Region 10 Guidance for Pacific Northwest State and Tribal Temperature Water Quality Standards. EPA 910-B-03-002. USEPA Region 10 Office of Water, Seattle, WA.

U.S. Environmental Protection Agency (EPA). 2017. Water Quality Standards Handbook: Chapter 3: Water Quality Criteria. EPA-823-B-17-001. EPA Office of Water, Office of Science and Technology, Washington, DC. Accessed February 2019. https://www.epa.gov/sites/production/files/2014-10/documents/handbook-chapter3.pdf.

USEPA. 2017c. Biological Evaluation for Reissuance of the NPDES Permit for the City of Sandpoint Wastewater Treatment Plant. NPDES Permit No. ID0020842. U.S. Environmental Protection Agency Region 10, Seattle, Washington. June 2017. 39 pp.

USFWS (U.S. Fish and Wildlife Service). 1998. Determination of threatened status for the Klamath River and Columbia River distinct population segments of bull trout. Federal Register Vol. 63 31647-31674.

USFWS (U.S. Fish and Wildlife Service). 2002. Bull trout (*Salvelinus confluentus*) draft recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon, October, 2002

USFWS. 2015. Recovery Plan for the Coterminous United States Population of Bull Trout (*Salvelinus confluentus*). Pacific Region U.S. Fish and Wildlife Service. Portland, Oregon. Available at: https://www.fws.gov/pacific/bulltrout/Planning.html. 195 pp.

U.S Fish and Wildlife Service Biological Opinion/Letter of Concurrence/Conference Concurrence Regarding U.S. Environmental Protection Agency's Proposed Approval of Revised Oregon Water Quality Standards for Temperature and Intergravel Dissolved Oxygen. July 6, 2015. 01EOFWO0-2014-F-0087. 303pp.

USFWS. 2015. Biological Opinion for the Environmental Protection Agency (EPA) on Oregon Water Quality Criteria for Toxic Chemicals. USFWS. Portland, OR.

Waples, R. S., R. P. Jones Jr., B. R. Beckman, and G. A. Swan. 1991. Status Review for Snake River Fall Chinook Salmon. NOAA Technical Memorandum NMFS F/NWS F/NWC-201.

WDFW (Washington Department of Fish and Wildlife). 1997. Washington State Salmonid Stock Inventory - Bull Trout/Dolly Varden. 444 pp.

WDFW/ODFW 2018 Joint Staff Report: Stock Status And Fisheries For Fall Chinook Salmon, Coho Salmon, Chum Salmon, Summer Steelhead, And White Sturgeon. Joint Columbia River Management Staff. Washington Department of Fish and Wildlife and Oregon Department of Fish and Wildlife. July 17, 2018. 74 pp,

Welch, E.B. 1980. Ecological Effects of Waste Water. Cambridge University Press. Cambridge.

Whitemore, C.M., C.F. Warren, and P. Doudoroff. 1960. Avoidance Reactions of Salmonid and Centrarchid Fishes to Low Oxygen Concentrations. Trans. Am. Fish. Soc. 89:17-26.

Wissmar, R., J. Smith, B. McIntosh, H. Li, G. Reeves, and J. Sedell. 1994. A history of resource use and disturbance in riverine basins of eastern Oregon and Washington (early 1800s-1990s). Northwest Science 68:1-35.

NOAA Fisheries (NMFS). 2015. Endangered Species Act Biological Opinion on the Environmental Protection Agency's Proposed Approval of Certain Oregon Water Quality Standards Including Temperature and Intergravel Dissolved Oxygen.

Zimmer, P. D. 1950. A Three-Year Study of Fall Chinook Salmon Spawning Areas in the Snake River Above Hells Canyon Dam Site. Supplements in 1951 and 1953. U.S. Fish and Wildlife Service Report.

10. APPENDICES

10.1. Appendix A. Memo from Peter Leinenbach, USEPA, describing data analysis for Snake River Water Temperature.

Methods – Snake River water temperature data was obtained from Idaho Power on January 4, 2018. Maximum weekly (7-day average) maximum temperatures (MWMT) were calculated from this data. Two data checks were implemented on this water temperature data: 1) The data was checked for "missing" data, with only MWMT estimates calculated with at least 4 days of data included in the analysis; 2) In addition, sampling data was removed from the analysis if more than 4 days of data were missing during the first week of the assessment period. The second check was done because water temperatures was almost always warmest during the initial part of the assessment period. Results of this analysis are presented in Tables 1 and 3.

Imnaha River, Salmon River and Clearwater River water temperatures were obtained from USFS NorWeST database (www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html#). Sampling locations used in this tributary analysis were the most downstream sampling location with water temperature data during the analysis period. Similar data checks were done to these datasets. The estimated Snake River RM (River Mile) associated with these tributary confluences was estimated from the NHDPlusV1 stream database (www.horizon-systems.com/NHDPlus/NHDPlusV1_home.php). Results of this analysis are presented in Tables 2 and 4.

Table 1. Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)[17]					
Year	RM 115.8 RB Logger	RM 115.9 RB Logger	RM 131.3 LB Logger	RM 131.4 LB Logger	RM 156.6 RB Logger
1991	- -	- -	- -	- -	18.4
1992	- -	- -	- -	- -	17.8
1993	- -	- -	- -	- -	17.7
1994	- -	- -	- -	- -	17.5
1995	- -	- -	- -	- -	16.8
1996	- -	- -	- -	- -	17.5
1997	- -	- -	- -	- -	16.2
1998	- -	- -	- -	- -	17.5
1999	- -	- -	- -	- -	16.7
2000	- -	- -	- -	- -	16.2
2001	- -	- -	- -	- -	18.8
2002	- -	- -	- -	- -	17.1
2003	- -	- -	- -	- -	19.5
2004	- -	- -	- -	- -	18.1
2005	- -	- -	- -	- -	16.7
2006	- -	- -	- -	- -	18.0
2007	- -	- -	- -	- -	16.1
2008	- -	- -	- -	- -	18.3
2009	- -	- -	- -	- -	17.2
2010	- -	- -	- -	- -	16.8
2011	- -	- -	- -	- -	19.2
2012	18.0	17.9	17.6	16.9	17.8
2013	15.7	15.6	14.9	14.5	15.5
2014	18.4	17.0	17.3	- -	18.3
2015	18.0	17.7	17.8	- -	- -
2016	18.1	17.0	- -	- -	- -
2017	17.3	17.2	- -	- -	17.0
2018	- -	- -	- -	- -	- -
Average	17.6	17.1	16.9	15.7	17.5
25th Percentile	17.5	17.0	16.7	15.1	16.8
75th Percentile	18.0	17.6	17.6	16.3	18.1

[17] Sampling locations are identified by river miles (RM), which corresponds with USGS quad map river mile designations and were provided by Idaho Power. The specific sample locations are further defined by channel cross section locations (i.e., RB, RC, LC, PS1), which corresponds with the instrument attached on Right Bank, Right half of the river channel, Left half of the river channel, Left Bank, and the Penstock of the Snake River Dam, respectively. The file name also indicate the type of instrument used to collect the data (i.e., Logger, Sonde, Gage), and sample site replicates are indicated by "1" in the this instrument name (i.e., RM 156.6 RB Logger1).

Count	6	6	4	2	25

Table 1 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)					
Year	RM 156.6 RB Logger1	RM 165.7 RB Logger	RM 165.75 RC Sonde	RM 169.7 RB Logger	RM 180.3 RB Logger
1991	- -	18.4	- -	18.4	- -
1992	- -		- -	17.9	- -
1993	- -	17.9	- -	16.1	17.7
1994	- -	17.9	- -	17.9	17.4
1995	- -	16.8	- -	16.9	17.3
1996	- -	17.4	- -	14.4	17.2
1997	- -	17.6	- -	- -	18.0
1998	- -	17.0	- -	- -	17.3
1999	- -	16.8	- -	- -	16.8
2000	- -	16.7	16.7	16.8	16.7
2001	- -	18.9	- -	19.1	18.8
2002	- -	17.0	- -	17.1	- -
2003	19.4	19.5	- -	19.4	19.2
2004	18.0	17.9	- -	17.9	18.0
2005	16.7	16.7	- -	16.9	16.9
2006	18.1	18.0	- -	18.2	18.0
2007	16.2	16.0	- -	16.3	16.3
2008	18.4	18.2	- -	18.3	18.3
2009	17.2	17.3	- -	- -	17.8
2010	- -	19.0	- -	- -	18.8
2011	19.2	19.2	- -	19.3	19.3
2012	- -	17.9	- -	18.0	18.1
2013	- -	15.5	- -	15.8	15.8
2014	- -	18.3	- -	18.3	18.2
2015	- -	18.8	- -	18.8	18.9
2016	- -	18.0	- -	18.0	18.1
2017	- -	17.1	- -	17.1	17.2
2018	- -	- -	- -	- -	- -
Average	17.9	17.7	16.7	17.6	17.8
25th Percentile	17.1	17.0	16.7	16.9	17.2
75th Percentile	18.6	18.3	16.7	18.3	18.2
Count	8	26	1	22	24

Table 1 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)					
Year	RM 180.3 RB Logger1	RM 189.0 LB Logger	RM 189.4 LC Sonde	RM 189.7 RB Logger	RM 192.3 RB Logger
1991	- -	19.2	- -	18.3	19.2
1992	- -	18.7	- -	17.9	18.6
1993	- -	18.2	- -	- -	18.2
1994	- -	18.9	- -	- -	- -
1995	- -	18.4	- -	- -	18.1
1996	- -	- -	- -	- -	17.4
1997	- -	18.3	- -	- -	- -
1998	- -	19.7	- -	- -	- -
1999	- -	- -	- -	- -	18.2
2000	- -	- -	17.6	- -	17.8
2001	- -	19.5	- -	- -	19.5
2002	- -	18.1	- -	- -	18.2
2003	19.3	19.8	- -	- -	19.8
2004	17.9	19.1	- -	- -	19.4
2005	16.9	18.0	- -	- -	18.1
2006	18.0	18.8	- -	- -	18.9
2007	16.3	17.8	- -	- -	18.0
2008	18.3	19.1	- -	- -	19.3
2009	17.8	19.0	- -	- -	19.2
2010	18.9	19.4	- -	- -	19.4
2011	19.4	20.0	- -	- -	20.0
2012		19.1	- -	- -	19.1
2013	15.9	18.7	- -	- -	- -
2014		19.7	- -	- -	- -
2015		19.7	- -	- -	- -
2016	18.1	18.7	- -	- -	- -
2017	17.2	18.5	- -	- -	- -
2018	- -	- -	- -	- -	- -
Average	17.8	18.9	17.6	18.1	18.8
25th Percentile	17.2	18.4	17.6	18.0	18.2
75th Percentile	18.4	19.4	17.6	18.2	19.3
Count	12	24	1	2	19

Table 1 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)					
Year	RM 202.3 LB Logger	RM 202.3 LB Logger1	RM 216.3 LB Logger	RM 227.5 LC Sonde	RM 229.7 LB Gage
1991	19.3	- -	19.4	- -	- -
1992	18.8	- -	18.8	- -	- -
1993	18.2	- -	18.1	- -	- -
1994	18.9	- -	19.1	- -	- -
1995	18.8	- -	18.8	- -	- -
1996	18.2	- -	18.2	- -	- -
1997	18.7	- -	- -	- -	- -
1998	- -	19.8	- -	- -	- -
1999	- -	19.2	- -	18.4	- -
2000	17.9	- -	17.8	17.8	- -
2001	19.6	- -	19.7	- -	- -
2002	18.2	- -	18.2	- -	- -
2003	19.7	18.3	- -	- -	- -
2004	19.3	18.9	19.4	- -	- -
2005	18.4	18.0	18.3	- -	- -
2006	18.8	19.3	18.9	- -	- -
2007	18.0	19.3	18.2	- -	- -
2008	19.4	19.4	19.3	- -	- -
2009	19.3	20.0	19.4	- -	- -
2010	19.5	18.8	19.5	- -	- -
2011	20.0	18.4	19.9	- -	- -
2012	19.2	- -	19.4	- -	- -
2013	19.0	- -	19.2	- -	- -
2014	19.7	- -	19.8	- -	20.2
2015	19.6	- -	19.8	- -	20.5
2016	18.8	18.8	18.9	- -	19.4
2017	18.4	18.4	18.4	- -	18.1
2018	- -	- -	- -	- -	- -
Average	18.9	19.0	19.0	18.1	19.6
25th Percentile	18.4	18.4	18.3	17.9	19.1
75th Percentile	19.4	19.3	19.4	18.2	20.3
Count	25	13	23	2	4

Table 1 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)					
Year	RM 229.8 LB Logger	RM 229.8 LB Logger1	RM 238.5 LC Sonde	RM 238.6 RB Logger	RM 241.3 RC Sonde
1991	19.5	- -	- -	19.5	- -
1992	18.8	- -	- -	18.7	- -
1993	18.0	- -	- -	- -	- -
1994	18.8	- -	- -	19.2	- -
1995	18.7	- -	- -	19.3	- -
1996	18.1	- -	- -	- -	- -
1997	17.3	- -	- -	- -	- -
1998	18.7	- -	- -	- -	- -
1999	- -	- -	- -	- -	- -
2000	17.9	- -	17.8		17.6
2001	19.9	- -	- -	- -	- -
2002	18.3	- -	- -	- -	- -
2003	19.7	- -	- -	- -	- -
2004	19.4	- -	- -	- -	- -
2005	18.3	- -	- -	- -	- -
2006	19.0	- -	- -	- -	- -
2007	18.4	- -	- -	- -	- -
2008	19.3	19.2	- -	- -	- -
2009	19.5	19.5	- -	- -	- -
2010	19.5	19.4	- -	- -	- -
2011	19.9	19.9	- -	- -	- -
2012	19.5	- -	- -	- -	- -
2013	19.2	- -	- -	- -	- -
2014	20.0	- -	- -	- -	- -
2015	19.8	- -	- -	- -	- -
2016	- -	- -	- -	- -	- -
2017	18.3	18.4	- -	- -	- -
2018	18.9	18.9	- -	- -	- -
Average	18.9	19.2	17.8	19.2	17.6
25th Percentile	18.3	19.0	17.8	19.1	17.6
75th Percentile	19.5	19.5	17.8	19.3	17.6
Count	26	6	1	4	1

Table 1 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)			
Year	RM 244.3 LB Sonde	RM 247.6 LC Sonde	RM 247.6 Penstock
1991	- -	- -	19.7
1992	- -	18.9	- -
1993	- -	- -	17.8
1994	- -	- -	19.6
1995	- -	- -	19.2
1996	- -	- -	18.2
1997	- -	- -	18.8
1998	- -	- -	19.6
1999	- -	- -	18.3
2000	- -	- -	18.0
2001	- -	- -	- -
2002	- -	- -	18.4
2003	- -	- -	19.5
2004	- -	- -	19.2
2005	- -	- -	18.4
2006	- -	- -	18.7
2007	- -	- -	18.4
2008	19.0	- -	19.3
2009	19.9	- -	19.6
2010	- -	- -	19.4
2011	- -	- -	19.8
2012	- -	- -	19.7
2013	- -	- -	19.5
2014	- -	- -	19.9
2015	- -	- -	19.9
2016	- -	- -	19.1
2017	- -	- -	18.3
2018	- -	- -	19.2
Average	19.5	18.9	19.1
25th Percentile	19.3	18.9	18.4
75th Percentile	19.7	18.9	19.6
Count	2	1	26

Table 2. Observed Clearwater River and Salmon River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 1st through November 14th (*C)[18]			
Year	Clearwater River at Mouth (Perma_FID 10692)	Salmon River at Mouth (Perma_FID 11094)	Imanaha River at Mouth (Perma_FID 7641 and 11811)
1991	- -	- -	14.1
1992	- -	- -	14.5
1993	- -	- -	- -
1994	13.9	- -	- -
1995	12.6	- -	- -
1996	15.6	- -	- -
1997	13.1	- -	- -
1998	13.4	- -	- -
1999	12.4	- -	- -
2000	12.0	- -	- -
2001	13.0	- -	- -
2002	12.9	- -	- -
2003	14.1	- -	- -
2004	13.4	15.2	16.1
2005	11.6	13.1	14.5
2006	13.9	14.8	15.6
2007	12.2	11.9	13.9
2008	14.5	15.0	15.9
2009	12.0	12.7	- -
2010	15.4	16.7	- -
2011	14.1	- -	- -
2012	- -	- -	- -
2013	- -	- -	- -
2014	- -	- -	- -
2015	- -	- -	- -
2016	- -	- -	- -
2017	- -	- -	- -
2018	- -	- -	- -
Average	13.3	14.2	15.1
25th Percentile	12.5	12.9	14.5
75th Percentile	14.0	15.1	15.9
Count	18	7	6

[18] Water temperature data was obtained from the United States Forest Service NorWeST database (www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html#). Sampling locations are the most downstream location with water temperature data during the analysis period, and the unique identifier in the NorWeST database associated with these sites is the Perma_FID value.

Table 3. Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)[19]					
Year	RM 115.8 RB Logger	RM 115.9 RB Logger	RM 131.3 LB Logger	RM 131.4 LB Logger	RM 156.6 RB Logger
1991	- -	- -	- -	- -	13.4
1992	- -	- -	- -	- -	- -
1993	- -	- -	- -	- -	- -
1994	- -	- -	- -	- -	13.6
1995	- -	- -	- -	- -	11.6
1996	- -	- -	- -	- -	11.5
1997	- -	- -	- -	- -	11.3
1998	- -	- -	- -	- -	12.1
1999	- -	- -	- -	- -	12.6
2000	- -	- -	- -	- -	12.0
2001	- -	- -	- -	- -	13.1
2002	- -	- -	- -	- -	12.2
2003	- -	- -	- -	- -	14.6
2004	- -	- -	- -	- -	13.0
2005	- -	- -	- -	- -	14.1
2006	- -	- -	- -	- -	12.5
2007	- -	- -	- -	- -	- -
2008	- -	- -	- -	- -	12.4
2009	- -	- -	- -	- -	12.3
2010	- -	- -	- -	- -	13.4
2011	- -	- -	- -	- -	13.4
2012	12.7	12.7	11.6	11.5	12.3
2013	12.1	12.1	11.9	11.8	12.4
2014	14.4		13.9	- -	14.4
2015	15.6	15.6	15.5	- -	- -
2016	12.8	12.6	- -	- -	- -
2017	11.9	11.8	- -	- -	11.9
2018	- -	- -	- -	- -	- -
Average	13.3	12.9	13.2	11.7	12.7
25th Percentile	12.2	12.1	11.9	11.6	12.1
75th Percentile	14.0	12.7	14.3	11.7	13.4

[19] Sampling locations are identified by river miles (RM), which corresponds with USGS quad map river mile designations and were provided by Idaho Power. The specific sample locations are further defined by channel cross section locations (i.e., RB, RC, LC, PS1), which corresponds with the instrument attached on Right Bank, Right half of the river channel, Left half of the river channel, Left Bank, and the Penstock of the Snake River Dam, respectively. The file name also indicate the type of instrument used to collect the data (i.e., Logger, Sonde, Gage), and sample site replicates are indicated by "1" in the this instrument name (i.e., RM 156.6 RB Logger1).

Count	6	5	4	2	22

Table 3 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)					
Year	RM 156.6 RB Logger1	RM 165.7 RB Logger	RM 165.75 RC Sonde	RM 169.7 RB Logger	RM 180.3 RB Logger
1991	- -	13.3	- -	13.5	- -
1992	- -	- -	- -	14.3	- -
1993	- -	12.9	- -	12.8	13.0
1994	- -	13.8	- -	13.9	13.9
1995	- -	11.5	- -	11.7	12.0
1996	- -	11.3	- -	11.9	11.8
1997	- -	11.2	- -	- -	11.6
1998	- -	12.0	- -	- -	12.2
1999	- -	12.7	- -	- -	- -
2000	- -	12.3	12.4	12.6	12.7
2001	- -	13.2	- -	13.5	13.5
2002	- -	12.3	- -	12.5	- -
2003	14.6	14.7	- -	14.8	14.8
2004	12.9	13.0	- -	13.2	13.4
2005	14.0	14.0	- -	14.1	14.1
2006	12.6	12.5	- -	12.8	12.7
2007	- -	12.0	- -	12.1	12.2
2008	12.4	12.5	- -	12.6	12.6
2009	12.3	12.3	- -	- -	12.6
2010	13.3	13.4	- -	0.0	13.6
2011	13.4	13.4	- -	13.6	13.6
2012	- -	12.3	- -	12.5	12.5
2013	- -	12.3	- -	12.4	12.4
2014	- -	14.3	- -	14.4	14.4
2015	- -	15.5	- -	15.7	15.8
2016	- -	13.7	- -	13.8	13.9
2017	- -	11.9	- -	11.9	12.0
2018	- -	- -	- -	- -	- -
Average	13.2	12.9	12.4	12.6	13.1
25th Percentile	12.5	12.3	12.4	12.4	12.3
75th Percentile	13.6	13.4	12.4	13.8	13.8
Count	8	26	1	23	23

Table 3 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)

Year	RM 180.3 RB Logger1	RM 189.0 LB Logger	RM 189.4 LC Sonde	RM 189.7 RB Logger	RM 192.3 RB Logger
1991	- -	15.2	- -	13.6	15.4
1992	- -	15.6	- -	14.3	15.6
1993	- -	14.7	- -	- -	14.8
1994	- -	15.1	- -	- -	15.1
1995	- -	14.0	- -	- -	13.4
1996	- -	14.2	- -	- -	- -
1997	- -	12.8	- -	- -	- -
1998	- -	14.3	- -	- -	14.3
1999	- -	- -	- -	- -	- -
2000	- -	- -	14.6	- -	14.7
2001	- -	15.3	- -	- -	15.4
2002	- -	14.4	- -	- -	14.5
2003	14.9	16.4	- -	- -	16.5
2004	13.3	15.5	- -	- -	15.8
2005	14.1	15.4	- -	- -	15.6
2006	12.7	14.7	- -	- -	15.0
2007	12.2	14.4	- -	- -	14.6
2008	12.7	14.6	- -	- -	14.8
2009	12.6	14.2	- -	- -	14.4
2010	13.6	15.9	- -	- -	16.0
2011	13.6	15.0	- -	- -	15.0
2012	- -	14.9	- -	- -	15.0
2013	12.4	14.8	- -	- -	- -
2014	- -	16.5	- -	- -	- -
2015	- -	17.4	- -	- -	- -
2016	13.9	15.8	- -	- -	- -
2017	12.1	14.4	- -	- -	- -
2018	- -	- -	- -	- -	- -
Average	13.2	15.0	14.6	14.0	15.0
25th Percentile	12.6	14.4	14.6	13.8	14.7
75th Percentile	13.7	15.5	14.6	14.1	15.5
Count	12	25	1	2	19

Table 3 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)					
Year	RM 202.3 LB Logger	RM 202.3 LB Logger1	RM 216.3 LB Logger	RM 227.5 LC Sonde	RM 229.7 LB Gage
1991	15.5	- -	15.6	- -	- -
1992	15.7	- -	15.8	- -	- -
1993	14.8	- -	14.9	- -	- -
1994	15.2	- -	15.4	- -	- -
1995	14.1	- -	14.1	- -	- -
1996	14.5	- -	14.6	- -	- -
1997	13.3	- -	12.2	- -	- -
1998	10.6	- -	13.0	- -	- -
1999	- -	- -	- -	- -	- -
2000	14.8	- -	14.8	14.9	- -
2001	15.6	- -	15.6	- -	- -
2002	14.6	- -	14.7	- -	- -
2003	16.6	16.5		- -	- -
2004	15.9	- -	16.0	- -	- -
2005	- -	- -	15.7	- -	- -
2006	14.9	15.0	15.1	- -	- -
2007	14.5	14.4	14.5	- -	- -
2008	14.8	- -	14.9	- -	- -
2009	14.4	14.3	14.5	- -	- -
2010	16.1	16.1	16.2	- -	- -
2011	15.1	15.1	15.1	- -	- -
2012	15.1	- -	15.2	- -	- -
2013	14.8	- -	15.1	- -	- -
2014	16.7	- -	16.8	- -	17.0
2015	17.4	- -	17.6	- -	18.6
2016	15.8	15.8	15.8	- -	16.1
2017	14.4	14.4	14.4	- -	14.2
2018	- -	- -	- -	- -	14.0
Average	15.0	15.2	15.1	14.9	16.0
25th Percentile	14.5	14.4	14.6	14.9	14.2
75th Percentile	15.7	15.9	15.7	14.9	17.0
Count	25	8	25	1	5

Table 3 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)					
Year	RM 229.8 LB Logger	RM 229.8 LB Logger1	RM 238.5 LC Sonde	RM 238.6 RB Logger	RM 241.3 RC Sonde
1991	15.8	- -	- -	15.8	- -
1992	16.0	- -	- -	15.9	- -
1993	14.9	- -	- -	14.9	- -
1994	15.3	- -	- -	15.5	- -
1995	14.0	- -	- -	- -	- -
1996	14.7	- -	- -	- -	- -
1997	13.4	- -	- -	- -	- -
1998	14.2	- -	- -	- -	- -
1999	- -	- -	- -	- -	- -
2000	14.8	- -	14.9		14.3
2001	15.8	- -	- -	- -	- -
2002	14.9	- -	- -	- -	- -
2003	16.9	- -	- -	- -	- -
2004	16.1	- -	- -	- -	- -
2005	15.8	- -	- -	- -	- -
2006	15.3	- -	- -	- -	- -
2007	14.7	- -	- -	- -	- -
2008	15.0	- -	- -	- -	- -
2009	14.5	14.5	- -	- -	- -
2010	16.3	- -	- -	- -	- -
2011	15.1	15.2	- -	- -	- -
2012	15.3	- -	- -	- -	- -
2013	15.1	- -	- -	- -	- -
2014	16.8	- -	- -	- -	- -
2015	17.7	- -	- -	- -	- -
2016		- -	- -	- -	- -
2017	14.4	14.5	- -	- -	- -
2018	15.5	15.6	- -	- -	- -
Average	15.3	14.9	14.9	15.5	14.3
25th Percentile	14.7	14.5	14.9	15.4	14.3
75th Percentile	15.8	15.3	14.9	15.8	14.3
Count	26	4	1	4	1

Table 3 (Continued). Observed Snake River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)				
Year	RM 242.8 LB Sonde	RM 244.3 LB Sonde	RM 247.6 LC Sonde	RM 247.6 Penstock
1991	- -	- -	- -	16.4
1992	- -	- -	15.8	- -
1993	- -	- -	- -	15.7
1994	- -	- -	- -	15.5
1995	- -	- -	- -	14.6
1996	- -	- -	- -	14.8
1997	- -	- -	- -	13.3
1998	- -	- -	- -	14.0
1999	- -	- -	- -	14.5
2000	16.9	- -	- -	15.0
2001	- -	- -	- -	- -
2002	- -	- -	- -	15.3
2003	- -	- -	- -	16.8
2004	- -	- -	- -	16.3
2005	- -	- -	- -	15.7
2006	- -	- -	- -	15.3
2007	- -	- -	- -	14.5
2008	- -	17.6	- -	14.9
2009	- -	14.5	- -	14.6
2010	- -	- -	- -	16.8
2011	- -	- -	- -	15.4
2012	- -	- -	- -	15.8
2013	- -	- -	- -	15.3
2014	- -	- -	- -	17.2
2015	- -	- -	- -	17.9
2016	- -	- -	- -	15.0
2017	- -	- -	- -	14.4
2018	- -	- -	- -	15.5
Average	16.9	16.0	15.8	15.4
25th Percentile	16.9	15.3	15.8	14.7
75th Percentile	16.9	16.8	15.8	15.8
Count	1	2	1	26

Table 4. Observed Clearwater River and Salmon River Maximum Weekly (7-Day Average) Maximum Water Temperatures (MWMT) for October 23rd through November 6th (*C)[20]			
Year	Clearwater River at Mouth (Perma_FID 10692)	Salmon River at Mouth (Perma_FID 11094)	Imanaha River at Mouth (Perma_FID 7641 and 11811)
1991	- -	- -	10.7
1992	- -	- -	10.3
1993	- -	- -	- -
1994	10.5	- -	- -
1995	8.4	- -	- -
1996	9.9	- -	- -
1997	9.4	- -	- -
1998	10.2	- -	- -
1999	9.6	- -	- -
2000	8.7	- -	- -
2001	10.3	- -	- -
2002	9.5	- -	- -
2003	10.1	- -	- -
2004	8.9	- -	8.7
2005	10.2	10.8	10.9
2006	9.1	8.6	10.9
2007	9.6	8.9	10.6
2008	8.9	8.4	10.0
2009	9.5	9.6	- -
2010	9.7	- -	- -
2011	10.3	- -	- -
2012	- -	- -	- -
2013	- -	- -	- -
2014	- -	- -	- -
2015	- -	- -	- -
2016	- -	- -	- -
2017	- -	- -	- -
2018	- -	- -	- -
Average	9.6	9.3	10.2
25th Percentile	9.1	8.6	10.1
75th Percentile	10.2	9.6	10.8
Count	18	5	6

[20] Water temperature data was obtained from the United States Forest Service NorWeST database (www.fs.fed.us/rm/boise/AWAE/projects/NorWeST.html#). Sampling locations are the most downstream location with water temperature data during the analysis period, and the unique identifier in the NorWeST database associated with these sites is the Perma_FID value.

10.2. Appendix B. Memo from Peter Leinenbach, USEPA, describing data analysis for Snake River water temperature interannual variability and decline in temperatures.

Water temperature data was obtained from Idaho Power. Temperature data collected at River Mile 229.8 was used in the analysis (Lat - 45.462162, Long -116.556731). **Figure 1** illustrates the Daily Maximum temperatures observed during from September 1st through October 22nd during the 1991 through 2008 period. The data indicates that daily maximum temperatures have generally trended higher over time at this site. **Table 1** presents the calculated average daily maximum Snake River temperatures during the fall period observed at the River Mile 229.8 monitoring station during this decade. The average daily temperature reduction at this site for both the October 9th through October 22nd and the October 9th through November 6th periods was 0.2*C.

Figure 1. Daily Maximum Water Temperatures in the Snake River from 1991 through 2018.

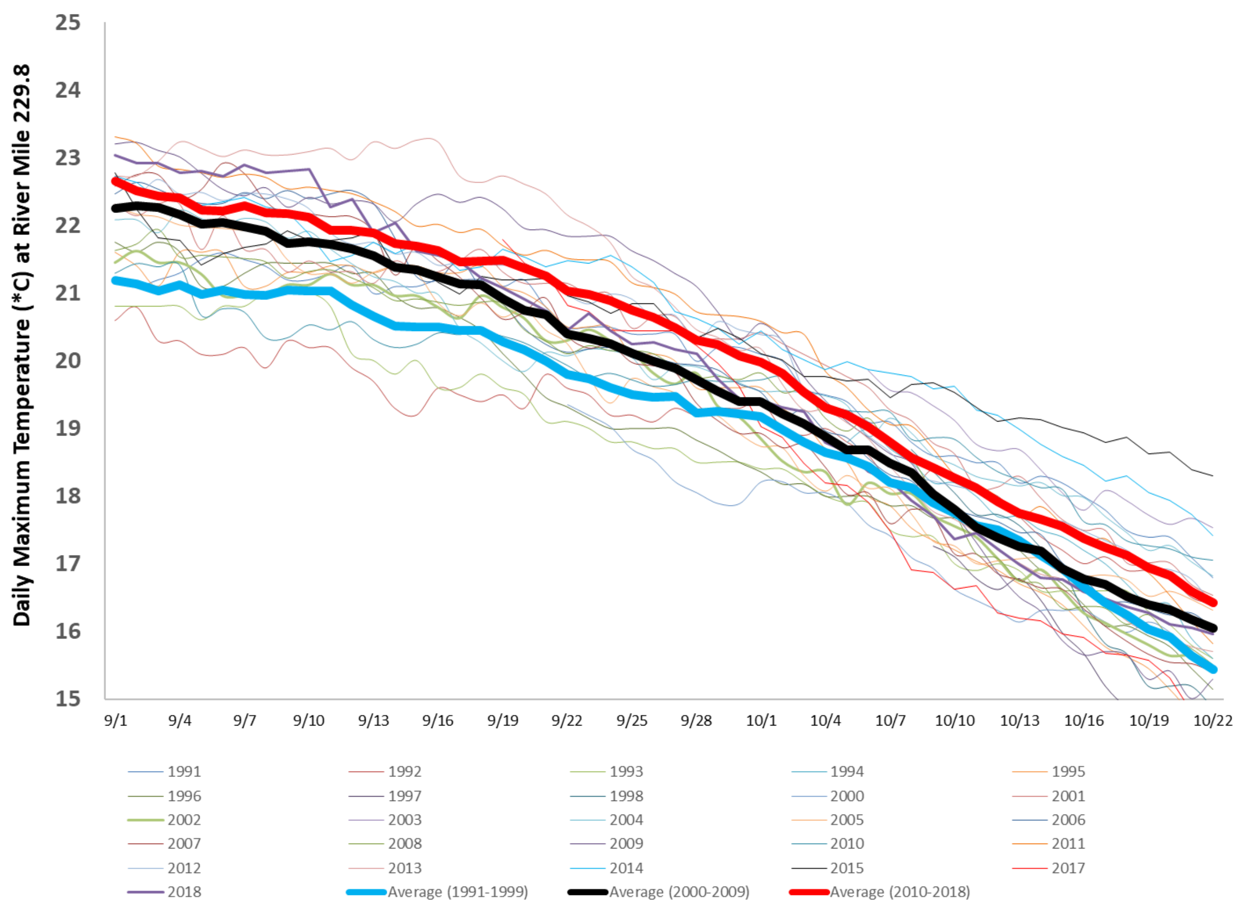
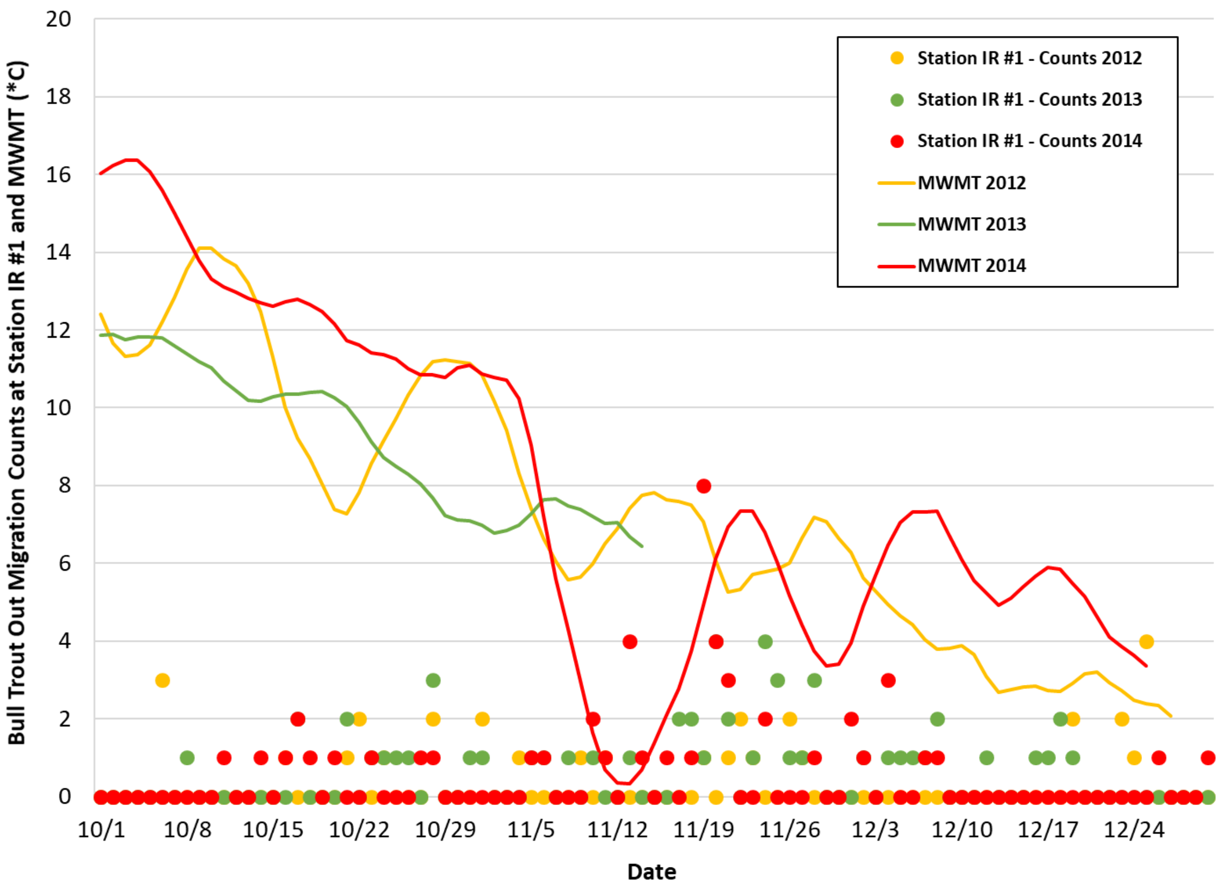


Table 1. Calculated Average Daily Maximum Snake River Temperatures During the Fall Period Observed at the River Mile 229.8 Monitoring Station During this Decade		
	Assessment Period	
Year	September 1st through October 22nd	October 9th through October 22nd
2010	19.5	17.9
2011	20.4	17.4
2012	20.1	17.5
2013	20.4	16.7
2014	20.6	18.6
2015	20.5	19.0
2016	No Data	No Data
2017	Incomplete Data	15.9
2018	19.8	16.7

10.3. Appendix C: Imnaha Bull Trout Outmigration Times and Imnaha River Temperatures, Memorandum from P. Leinenbach, 2019.

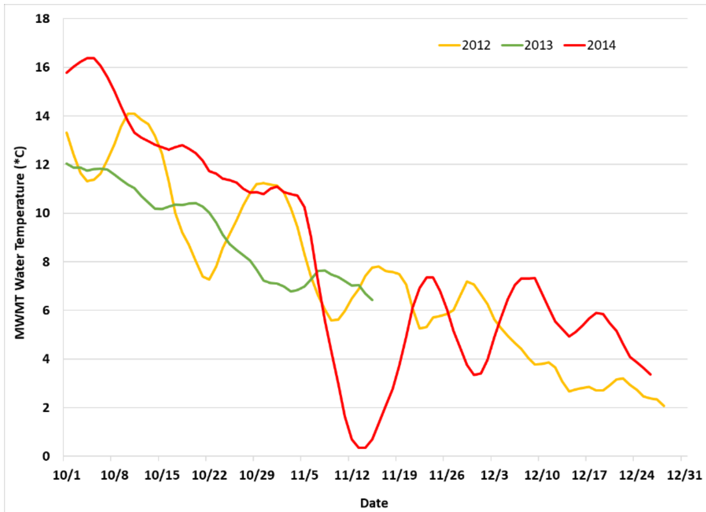
Hourly Imnaha River water temperature data, along with daily Bull Trout downstream migration counts at the most downstream site within the Imnaha River (IR#1), was obtained from Idaho Power. The total number of new daily Bull Trout downstream migration counts and calculated Maximum Weekly (7-Day Average) Maximum Water Temperature (MWMT) statistics for the fall period in 2012 through 2014 are presented in **Figure 1**.

Figure 1. The total number of new daily Bull Trout downstream migration counts out of the Imnaha River and calculated Maximum Weekly (7-Day Average) Maximum Water Temperature (MWMT) statistics in the Imnaha River for the fall period of 2012 through 2014.



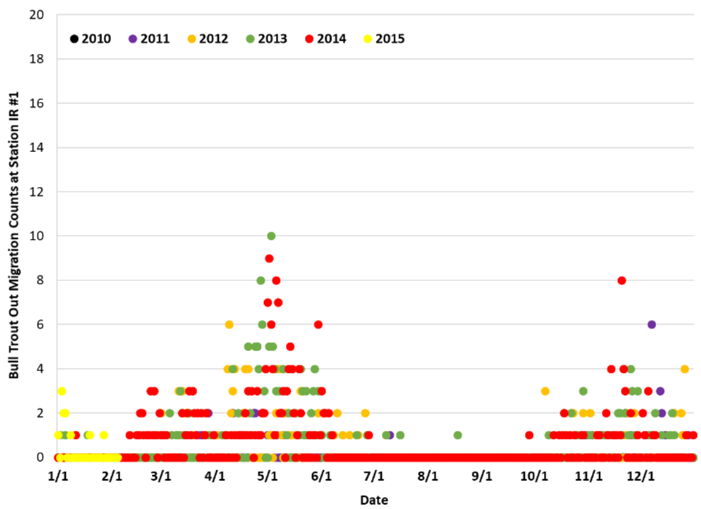
Methods – Fall period (i.e., post October 1st) MWMT statistics were estimated for the Imnaha River from hourly temperature data provided to EPA from Idaho Power (**Figure 2**). Please note that temperature data was not available for the entire 2011, and part of fall period of 2013.

Figure 2. Calculated MWMT Statistics for the Imnaha River.



The total number of new daily Bull Trout migration counts were plotted for the Imnaha River for the 2010 through 2015 period (**Figure 3**). There are two distinct elevated count periods – spring and fall. The spring period represents migration upstream into the Imnaha River and the fall period represents migration downstream out of the Imnaha River. Please note that there is a full season of spawning data for 2011 (i.e., purple dots in **Figure 3**), however there was no temperature data collected during this period. In addition, please note that Bull Trout downstream migration data (i.e., Fall Data) was only collected for a brief period in 2010 and not at all in 2015. Accordingly, a comparison between the MWMT statistic (**see Figure 2**) and migration counts (**see Figure 3**) were implemented for the fall period when both datasets were available (e.g., 2012, 2013, and 2014) (**see Figure 1**).

Figure 3. The total number of new daily Bull Trout migration counts in the Imnaha River (Site IR#1) for the 2010 through 2015 period.



10.4. Appendix D: Hanford Reach Columbia River Temperature and fall-run Chinook Spawning Patterns Memorandum from Ritchie Graves transmitted to Dan Opalski, 2019

[see attached pdf]

10.5. Appendix E: Research, Monitoring, and Evaluation of Emerging Issues and Measures to Recover the Snake River Fall Chinook Salmon ESU (Connor et al. 2017)

[attached as pdf]

UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
West Coast Region
1201 NE Lloyd Boulevard, Suite 1100
PORTLAND, OREGON 97232-1274

February 8, 2019

Dan Opalski, Director
Office of Water and Watersheds
US EPA, Region 10
1200 Sixth Avenue, Suite 155
Seattle, WA 98101

Re: Hanford Reach Fall Chinook Spawning Counts and Associated Temperature Data

Dear Dan Opalski,

The Environmental Protection Agency (EPA) is considering an increase to the spawning temperature standard (7-day average of the daily maximum) from 13.0 degrees Celsius to 14.5 degrees Celsius. The Snake River fall Chinook ESU (Evolutionary Significant Unit), listed as threatened under the Endangered Species Act, is the species most likely to be affected by this change. Snake River fall Chinook are closely related to the Upper Columbia River summer/fall Chinook, which are not listed under the ESA. The Hanford Reach fall Chinook population is part of this Upper Columbia ESU and is one of the most productive Chinook salmon populations on the West Coast of the United States. The Columbia Basin Hydropower Branch at the NOAA Fisheries West Coast Regional Office in Portland has summarized data in the attached document which includes Hanford Reach (Upper Columbia River) fall Chinook spawning redd counts and recorded water temperatures for the same dates and location. We believe this information may be helpful in EPA's deliberations.

The data sources used in the summary include the following:

Temperature Data – The Army Corps of Engineers Data Query Website for Priest Rapids Temperature Gauges, specifically, hourly data collected by the gauge located on Vernita Bar Bridge (PRXW.Temp-Water.Inst.1Hour.0.GCPUD-RAW) available for years: 2004, 2006, 2008, 2009, 2012, 2013, 2014, 2015, and 2016.

http://www.nwd-wc.usace.army.mil/dd/common/dataquery/www/?k=priest%20rapids

Fall Chinook Spawning Data – Aerial redd count data (for the years which hourly temperature data were available) for the Vernita Bar reach, and the entire Hanford reach population, provided by Grant County PUD & WDFW staff. The maximum count for each year (not all dates) can also be found in annual "Priest Rapids Hatchery Monitoring and Evaluation Reports".

The Vernita Bar reach is the most productive spawning area for the Hanford Reach fall Chinook population, and is also closest in proximity to the Vernita Bar bridge temperature gauge, which is why we chose to use it as the primary location for our data analysis. The relationship between the spawning redd counts and 7-day averages of the daily maximum temperature at this location demonstrates that this population of Upper Columbia River summer-fall Chinook initiates spawning behavior at temperatures above or near 14.5 degrees Celsius and has done so since the year 2004, if not before this time period.

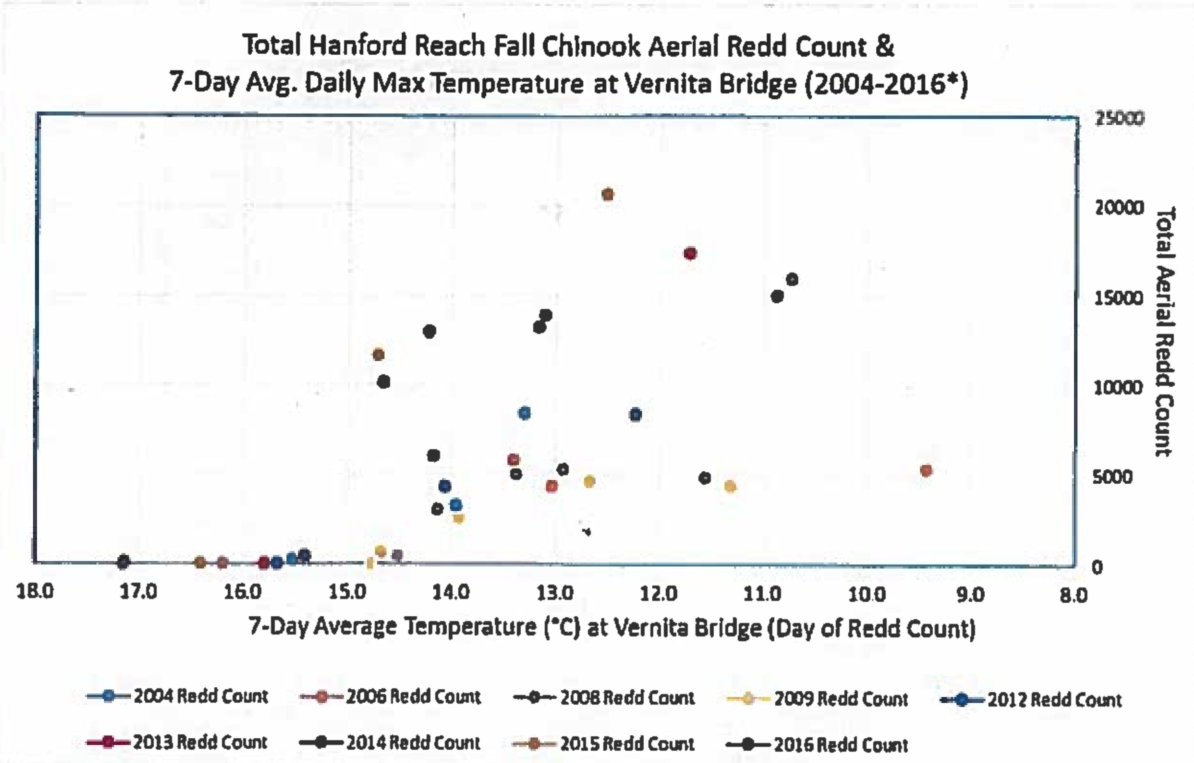
Please let us know if you have any questions about the information provided or the data sources.

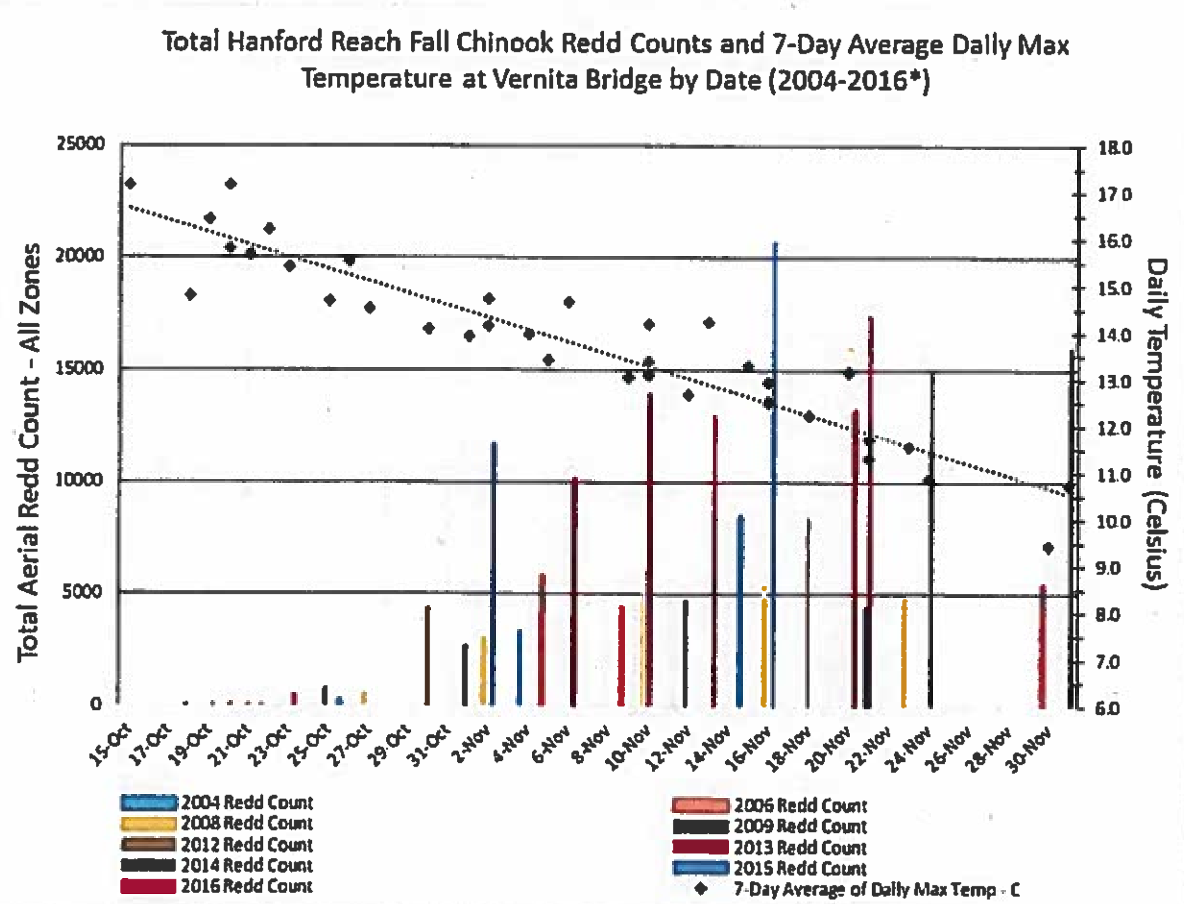
Sincerely,

Ritchie J Graves

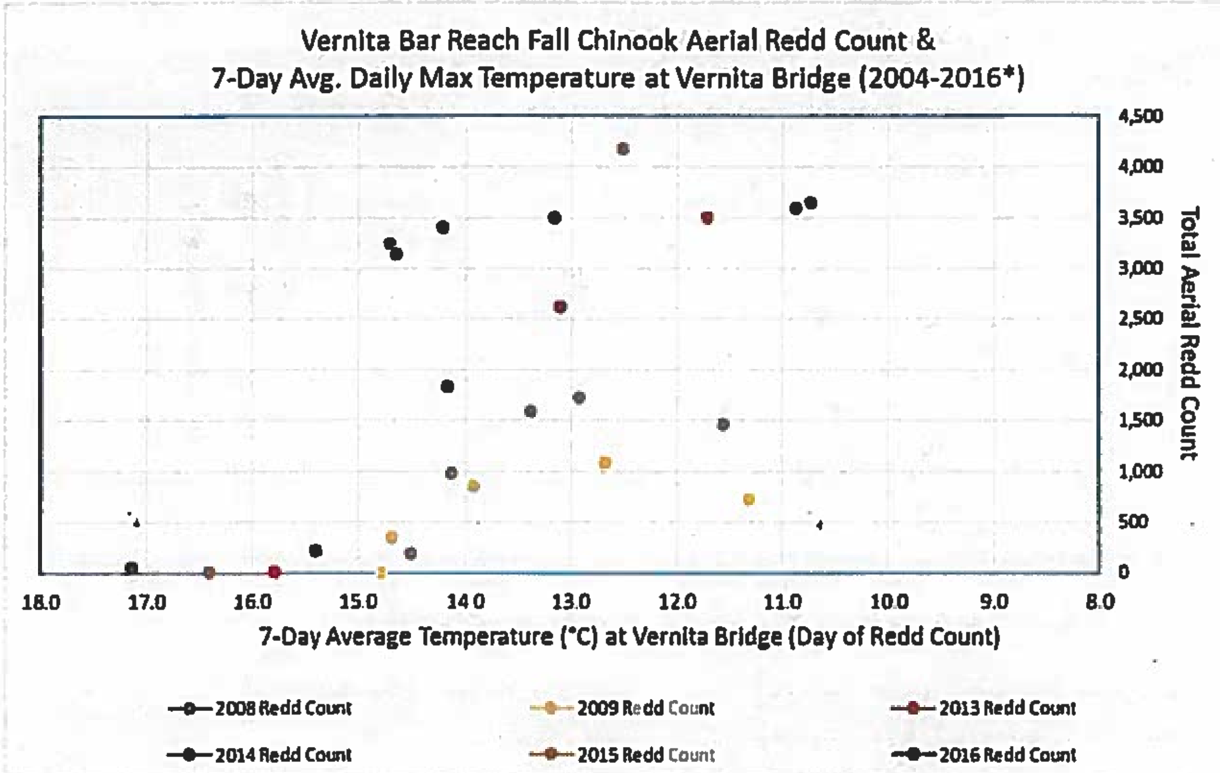
Ritchie J Graves, Chief
Columbia Hydropower Branch
Interior Columbia Basin Office
NOAA Fisheries, West Coast Region

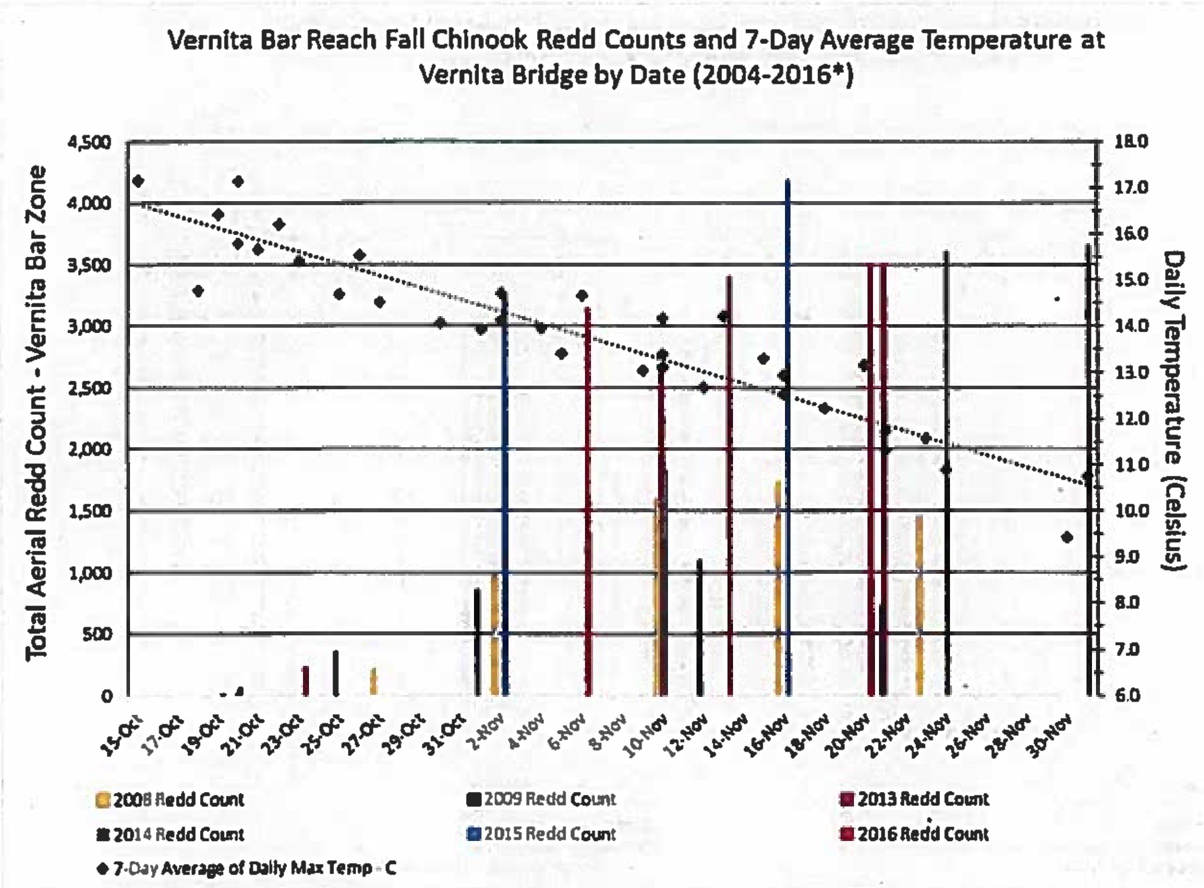
Cc: Rochelle Labiosa, EPA
John Palmer, EPA
Laurie Beale, EPA
Jennifer Byrne, EPA
Michael Tehan, NOAA





*Hourly temperature data from the Vernita Bar bridge gauge was not available in 2005, 2007, 2010, 2011 or 2017 for these dates.





*Aerial redd survey data specific to Vernita Bar reach was only available for a subset of the year in which hourly temperature data at Vernita Bar bridge was also available. Vernita Bar reach is the most productive reach for salmon spawning redds in the Hanford Reach population.

Research, Monitoring, and Evaluation of Emerging Issues and Measures to Recover the Snake River Fall Chinook Salmon ESU

BPA Project Number 199102900

Report covers work performed under BPA Contract # 72899, 72898

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Abstract

The portion of the Snake River fall Chinook Salmon *Oncorhynchus tshawytscha* ESU that spawns upstream of Lower Granite Dam transitioned from low to high abundance during 1992–2016 in association with U.S. Endangered Species Act recovery efforts and other federally mandated actions. This annual report focuses on (1) numeric and habitat use responses by natural- and hatchery-origin spawners, (2) phenotypic and numeric responses by natural-origin juveniles, and (3) predator responses in the Snake River upper and lower reaches as abundance of adult and juvenile fall Chinook Salmon increased. Spawners have located and used most of the available spawning habitat and that habitat is gradually approaching redd capacity. Timing of spawning and fry emergence has been relatively stable; whereas the timing of parr dispersal from riverine rearing habitat into Lower Granite Reservoir has become earlier as apparent abundance of juveniles has increased. Growth rate (g/d) and dispersal size of parr also declined as apparent abundance of juveniles increased. Passage timing of smolts from the two Snake River reaches has become earlier and downstream movement rate faster as estimated abundance of fall Chinook Salmon smolts in Lower Granite Reservoir has increased. In 2016, we described estimated the consumption rate and loss of subyearlings by Smallmouth Bass before, during, and after four hatchery releases. Before releases, Smallmouth Bass consumption rates of subyearling was low (0–0.36 fish/bass/d), but the day after the releases consumption rates reached as high as 1.6 fish/bass/d. Bass consumption in the upper portion of Hells Canyon was high for about 1–2 d before returning to pre-release levels, but in the lower river consumption rates were reduced but took longer to return to pre-release levels. We estimated that most of the subyearlings consumed by bass were of hatchery origin. Smallmouth Bass predation on subyearlings is intense following a hatchery release, but the predation pressure is relatively short-lived as subyearlings quickly disperse downstream. This information will allow us to better estimate subyearling loss to predation from our past efforts at time intervals less than 2 weeks. These findings coupled with stock-recruitment analyses presented in this report provide evidence for density-dependence in the Snake River reaches and in Lower Granite Reservoir that was influenced by the expansion of the recovery program. The long-term goal is to use the information covered here in a comprehensive modeling effort to conduct action effectiveness and uncertainty research and to inform Fish Population, Hydrosystem, Harvest, Hatchery, and Predation and Invasive Species Management RM&E.

Introduction

The ISAB (2015) wrote “Understanding density dependence—the relationship between population density and population growth rate—is important for effective implementation of the Columbia River Basin Fish and Wildlife Program, biological opinions, recovery plans, and tribal programs. Information on how density dependence limits fish population growth and habitat carrying capacity is vital for setting appropriate biological goals to aid in population recovery, sustain fisheries, and maintain a resilient ecosystem. Habitat restoration and population recovery actions can be planned and implemented more effectively by understanding mechanisms that cause density dependence in particular cases, such as limited food supply, limited rearing or spawning habitat, or altered predator-prey interactions.”

Management efforts have been implemented in response to listing under the Endangered Species Act (ESA; NMFS 1992) to increase the size of the population and survival of Snake River basin fall Chinook Salmon *Oncorhynchus tshawytscha* (e.g., reduced harvest, Peters et al. 2001; stable minimum spawning flows, Groves and Chandler 1999; summer flow augmentation, Connor et al. 2003b; predator control, Beamesderfer et al. 1996; increased hatchery production and supplementation; improved dam passage structures, Rainey et al. 2006; summer spill operations, CBR 2015). To track changes in attributes of the natural-origin population as abundance increased, Connor et al. (2013) divided the years 1991–2011 into periods of low and high abundance. To track changes in the attributes of spawning, rearing, emigration, and predation in this report, we added the years 2012 through 2016 to the period of record. The low and high abundance periods for adults were set at 1991–1998 and 1999–2016. The low and high abundance periods for juveniles were set at 1992–1999 and 2000–2016. Estimated escapement of natural- and hatchery-origin spawners upstream of Lower Granite Dam (hereafter, total escapement) increased markedly between the two abundance periods reaching a post-ESA listing high of 52,989 in 2014 and 52,338 in the most recent year 2016 (Table 1).

To assist with the monitoring of recovery measures, staff of project 199102900 have collected and analyzed data on adult and juvenile fall Chinook Salmon collected along the lower Snake River upper and lower reaches to the tailrace of Lower Granite Dam (Figure 1) since brood year 1991 (fry emergence year 1992). That project functions in the long-term as a research project by publishing papers to help to answer uncertainty and action effectiveness questions, while reporting interim information on status and trends as the data are collected. Predation by nonnative fishes is one factor that has been implicated in the decline of juvenile salmonids *Oncorhynchus* spp. in the Pacific Northwest, but it has been scantly studied in the case of Snake River fall Chinook Salmon. The only evaluation of predation on subyearling Snake River fall Chinook Salmon in the Snake River upper and lower reaches was conducted by Nelle (1999). Within the upper reach, Nelle (1999) reported that subyearlings only made up 1.9% and 0.8% of Smallmouth Bass diets by weight in 1996 and 1997, respectively. That study was conducted during the low abundance period soon after the Snake River fall Chinook Salmon ESU was listed under the ESA in 1992. Thus, low abundance of fall Chinook Salmon could explain why Smallmouth Bass consumption rates were relatively low compared to those from studies conducted in the Columbia and Yakima rivers where salmon abundance was higher (Tabor et al. 1993; Fritts and Pearsons 2004).

To date, our predation work has focused on estimating the loss of subyearling fall Chinook Salmon to Smallmouth Bass predation in Hells Canyon. Our methodology relies on estimating Smallmouth Bass consumption rate of subyearlings and then expanding that rate by bass abundance in a reach over a specific sampling interval, which was two weeks in our past studies. Thus an assumption of this method is that consumption rate does not vary over this time interval. However, the assumption is violated when large hatchery releases occur and bass consumption rate changes within the two-week interval. Without knowing how the consumption rate changes when a hatchery release occurs, estimates of subyearling loss can be either over or under estimated. In 2016, we attempted to better understand changes in bass consumption rates before, during, and after hatchery releases so as to correct our results when such events occur. An additional objective was to determine the number of hatchery fish consumed in our study area during each release.

TABLE 1. Estimates of escapement of natural- and hatchery-origin adult (≥ 53 cm FL) fall Chinook Salmon from the Snake River basin ESU, 1991–2016. Reference the following for the results: Busack (1991), Cooney (1991), LaVoy (1992, 1993, 1994, 1995), LaVoy and Mendel (1996), Mendel and LaVoy (1997), Mendel (1998, 1999, 2000), Young et al. (2012, unpublished). The mean (± SD) escapements for the abundance period estimates (Low, 1991–1998; High, 1999–2016) are also given.

Year	Natural	Hatchery	Total
1991	318	253	571
1992	549	111	660
1993	742	195	937
1994	406	186	592
1995	350	267	617
1996	639	260	899
1997	797	195	992
1998	306	610	916
1999	905	890	1,795
2000	1,148	1,410	2,558
2001	5,163	4,382	9,545
2002	2,116	7,231	9,347
2003	3,455	8,974	12,429
2004	2,637	9,773	12,410
2005	4,584	5,340	9,924
2006	3,984	2,501	6,485
2007	2,816	5,538	8,354
2008	2,995	8,930	11,925
2009	4,273	16,412	20,685
2010	7,347	32,417	39,764
2011	8,072	15,508	23,580
2012	11,315	19,048	30,363
2013	20,425	30,813	51,238
2014	13,141	39,848	52,989
2015	15,422	36,916	52,338
2016	8,762	23,110	31,872
Low	513 ± 183	260 ± 141	773 ± 167
High	6,587 ± 5,206	14,947 ± 12,182	21,533 ± 16,743

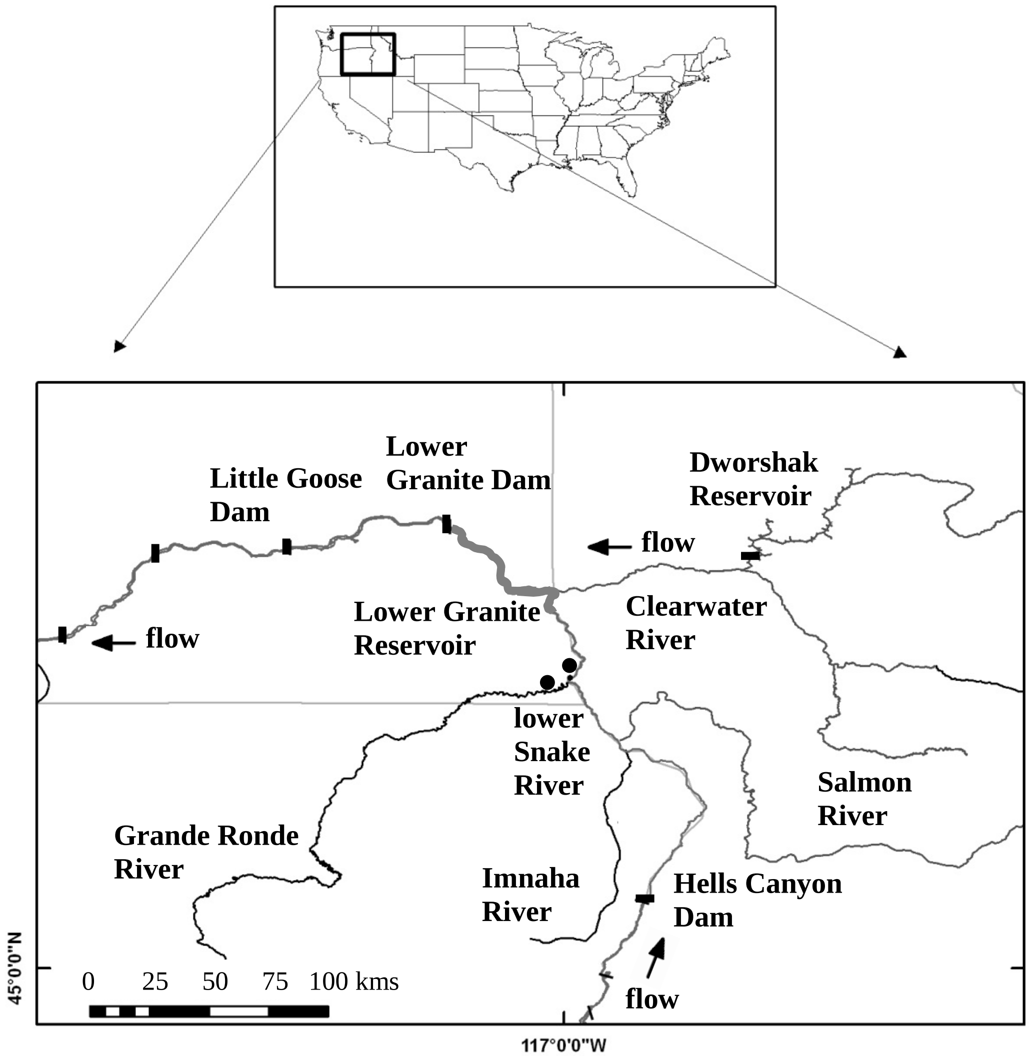

FIGURE 1. The Snake River basin including the free-flowing but regulated Snake River upper (Hells Canyon Dam to Salmon River mouth) and lower (Salmon River mouth to the upper end Lower Granite Reservoir) reaches where natural- and hatchery-origin adults spawned during 1991–2016 and natural-origin fall Chinook Salmon subyearlings were captured seined and PIT tagged while rearing during 1992–2016. Lower Granite Reservoir and Lower Granite Dam are the first of the impoundments and dams encountered by the fish after they had become smolts during early seaward migration.

The goal of this annual report is to describe how the status and trends in the data collected on fall Chinook Salmon in the Snake River upper (Hells Canyon Dam to the Salmon River mouth) and lower (Salmon River mouth to upper end of Lower Granite Reservoir; Figure 1) reaches provide evidence for density-dependent interactions including an update on the response to predation. The objectives of this report are to summarize information collected on attributes of (1) spawning in riverine habitat, (2) natural-origin juveniles rearing in riverine habitat, (3) natural-origin juveniles emigrating through Lower Granite Reservoir, and (4) seasonal variation in Smallmouth Bass diets and consumption of subyearling Chinook Salmon during rearing in riverine habitat.

Methods

Attributes of Spawning in Riverine Habitat (Protocol ID 2058, Published)

Aerial surveys were scheduled at 7-d to 14-d intervals starting in late October and ending in early December. The number of flights made varied by year. Redds were counted from a helicopter flown about 200 m above the river, which allows observing 100% of the river bottom at depths approximately < 3 m (i.e., shallow water). Beginning in 2015, aerial surveys were replaced with surveys conducted by staff of the Idaho Power Company by use of a small unmanned aircraft system (sUAS). For the sUAS survey samples, 35 individual sites were flown four times (every other week) during the spawning season, beginning the week of 10/26, and ending during the week of 12/07. The total number of new redds was compiled for each of the 35 sample sites, for each survey, and that provided a total number of redds present at each site that were incorporated into a sampling proportional to size model to estimate (95% C.I.) total redd counts for each Snake River reach. The efficacy of the sUAS surveys is described by Groves et al. 2016). Problems associated with sampling error in 2015 were resolved in 2016.

Potential deep water spawning locations (>3 m) were searched for redds as described in Protocol ID 2058 and by Groves et al. (2013). We tabulated the redd counts by year and reach. We calculated inter-annual mean redd counts by abundance period. To evaluate the evidence for density-dependence during spawning, we plotted the annual numbers of shallow redds counted (or estimated after 2014), number of shallow sites used (restricted to years prior to 2015), deep redds counted, and deep sites used against the corresponding brood year estimates of total escapement (Table 1). Plots were made for the reaches jointly. To assess the plausibility and strength of density-dependence during spawning, we assumed a log-normal error structure and fit three types of linear models. First, in the following manner we fit a simple linear regression model so that recruitment (*R*; e.g., aerial redd counts) increased proportionately and indefinitely (no density-dependence) with increases in total escapement (*S*):

(1) $$R_{t+1} = \theta_0 + \theta_1 S_t \, .$$

Secondly, we fit the Beverton-Holt model:

(2) $$R_{t+1} = \frac{\alpha S_t}{1+(\beta S_t)} \, ,$$

whereby α measures productivity when the number of spawning adults is near zero (i.e., density-independence), and β measures the strength of density-dependence as the number spawning adults increases. The Beverton-Holt model reaches an asymptote in the number recruits produced at α/β.

Lastly, we fit the Ricker production model expressed as:

(3) $$R_{t+1} = \rho S_t e^{-\delta S_t},$$

where ρ is similar to α in the Beverton-Holt and measures density-independent productivity (proportional to fecundity) and δ measures the strength of density-dependence. We used Akaike's Information Criterion (AIC) to pick the model with the best fit.

To evaluate the time of spawning, if redds were counted during a given aerial survey we assigned a spawning date to that survey by subtracting seven days from the flight date. For example, if the first flight was made on 10/21 and 10 redds were counted, the first spawning date would be 10/14. We calculated the first, peak, and last spawning dates as just described and calculated the percentages of the total redd count made on those dates. We calculated inter-annual means for the first, peak, and last spawning dates and for the percentages on those dates by abundance period. Analyses on time of spawning analyses were restricted to the years 1992–2014 because the flight schedules for the sUAS surveys that began in 2015 were shifted to slightly later dates and the resulting count data were not directly comparable to the data collected in prior years.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat (Protocol ID 2057, Published)

Attributes evaluated during rearing in riverine habitat included apparent abundance of natural-origin subyearlings, timing of fry (< 46-mm FL) presence, and timing of parr (> 45-mm FL; after Connor et al. 2002) dispersal from riverine habitat, parr dispersal size, and parr growth. To collect data on those attributes, we used a beach seine at 11–15 permanent stations located along 142 contiguous kms of the two riverine reaches studied. Large portions of the hatchery smolts released into the river were released without an external mark or fin clip (i.e., unmarked). After hatchery smolts were released upstream of a given seining station, the origin (i.e., natural or hatchery) of each collected unmarked fish was classified based on morphology (overall accuracy 98.7%; Tiffan and Connor 2011). During 1992–2007, we implanted each natural-origin parr ≥ 60 mm long collected during seining in riverine habitat with an 11.5-mm passive integrated transponder (PIT) tag (Prentice et al. 1990a). During 2008–2016, in addition to tagging fish longer than 59 mm with 11.5-mm tags, we also tagged natural-origin parr that were 50–59 mm long with 8.5-mm tags. Natural-origin parr were given 15 min to recover from tagging in an aerated 19-L bucket of river water before release at their capture site.

Mean daily CPUE (natural-origin subyearlings per seine haul) values for each sampling station were adjusted for the presence of natural-origin, spring/summer Chinook Salmon and averaged across sampling stations and weeks within a year to calculate mean annual CPUE (± SE) by reach and for reaches combined as an indices of apparent abundance in riverine rearing habitat, where *N* was the number of station visits. We also calculated mean inter-annual CPUE by abundance period (low abundance period, 1992–1999; high abundance period, 2000–2016; after Connor et al. 2013). To evaluate the evidence for density-dependence during rearing in riverine habitat, mean annual CPUE for the combined reaches was plotted against the corresponding brood year estimates of total escapement (Table 1) and curves were fit to the data as described for redd counts.

The median, minimum, and maximum day of year (January 1 = 1) of fry and parr presence were tabulated by reach and year. We calculated inter-annual means from the annual median dates of presence for each abundance period under the premise that dispersal timing into the reservoir became earlier as the median dates of parr presence became earlier. We calculated annual mean and abundance period wet weights (0.1 g) of all natural-origin parr captured in the riverine habitat reasoning that decreases in weight reflected decreases in size at dispersal (and vice versa). We calculated absolute growth rates (g/d) of individual PIT-tagged, natural-origin parr recaptured by beach seine as (WT2 – WT1) /(Day2 – Day1) and used those growth rates to calculate annual mean growth rates for fish recaptured within each year, and inter-annual mean growth rates for each abundance period.

Attributes of Natural-Origin Juveniles Emigrating through Lower Granite Reservoir (Protocol ID 2057,Published)

The basin-wide population of subyearling smolts consists of the aggregate of natural- and hatchery-origin fish produced or released upstream of Lower Granite Reservoir. We modified Method ID 3999 (Published) to estimate the daily number of natural-origin, fall Chinook Salmon subyearling smolts that passed Lower Granite Dam each year. We summed the daily estimates made for March through October to estimate seasonal passage abundance. To evaluate the evidence for density-dependence during early seaward migration in the reservoir, we plotted estimated annual passage abundance at the dam against the corresponding brood year estimates of total escapement (Table 1) and curves were fit to the data as described for redd counts..

We also used Method ID 3999 (Published) to estimate daily passage abundance of natural-origin subyearlings at Lower Granite Dam that had been PIT tagged in the Snake River reaches. The annual median passage dates for each reach were plotted against year, and the inter-annual means of those medians were calculated by abundance period. Downstream movement rate was calculated for individual, natural-origin subyearlings that had been PIT tagged while rearing in riverine reaches of the Snake River as the elapsed days between release and detection at a point downstream divided by the channel distance in river kms traversed between release and subsequent detection at Lower Granite Dam. Annual means, medians, minimums, and maximums were calculated by reach and abundance period using the downstream movement rates of individual fish.

Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat (Protocol ID 299, Published)

We examined the daily consumption and diet of Smallmouth Bass before and during four hatchery releases of subyearling fall Chinook Salmon within Hells Canyon, an unimpounded section of the Snake River (Figure 2). These releases occur annually within our study area. The first two hatchery releases we studied in 2016 were made at Hells Canyon Dam (rkm 398.5) during which 1,041,185 fish were released over 2 days (May 16 and 18; Table 2). A third release of 398,086 subyearlings was made on May 20 from the Pittsburg Landing acclimation facility (rkm 345.5). To evaluate these three releases, we first divided our study area into two reaches: one downstream of Pittsburg Landing from rkm 328.1 to 345.5, and one upstream from rkm 345.5 to 365.3. We began sampling in both reaches on May 15, the day before the first release at Hells Canyon Dam. Sampling continued in each reach until subyearlings were no longer observed (estimated in the field) in the diets of bass. The fourth hatchery release we studied was made on June 10 at the Captain John acclimation facility (rkm 262.8) during which 198,983 fish were released. We sampled the river from the acclimation facility downstream to the head of Lower Granite Reservoir at Asotin, WA (rkm 234.2). Sampling began the morning before the release and continued until the presence of subyearlings in bass stomachs were similar to prerelease levels.

Smallmouth Bass ≥150mm TL were collected by angling to evaluate consumption for the Hells Canyon Dam and Pittsburg Landing releases and by boat electrofishing to evaluate the Captain John release. Different gears were used to be consistent with the methods we used during our previous consumption studies. All sampling sites were randomly selected according to shoreline habitat type that was provided as a GIS habitat layer by the Idaho Power Company. The following shoreline habitat types were sampled: pool, riffle, glide, bar, and fan. We did not sample “rapids” because they represented a relatively small (<5%) amount of shoreline and were difficult to sample. Sites that were angled were sampled for ~20 min by 2 people fishing from a boat. Sites that were electrofished were sampled along 80- to 800-m transects that typically encompassed an entire specific habitat type. One netter collected fish.

Consumption

We calculated the consumption rate C (number of Chinook Salmon/Smallmouth Bass/day) in a series of steps similar to Fritts and Pearsons (2004). First, we identified ingested fish from bass stomachs using diagnostic bones (i.e., dentary, cleithrum, opercle; Parrish et al. 2006) and estimated their original FL and weight at ingestion using various regressions (Hansel et al. 1988; Vigg et al. 1991; Parrish et al. 2006). These weights were summed with weights of other diet items (if present) to derive a meal weight (MW) for each individual bass (Vigg et al. 1991). We accounted for a 21.3% weight loss associated with preservation in 90% ethanol for all diet items used in the calculation (Shields and Carlson 1996). Next, we input MW into an evacuation rate model of Smallmouth Bass digestion of salmonids developed by Rogers and Burley (1991) and modified by Fritts and Pearsons (2004) that predicts time (in hours) to 90% evacuation (ET_{90}):

$$ET_{90} = (24.542)(MW^{0.29}e^{-0.15T}W^{-0.23})(24),$$

where W is bass weight (g; measured or estimated), and T is temperature. Finally, we calculated C for each individual bass using the equation presented by Ward et al. (1995):

$$C = n(24/ET_{90}),$$

where n is the number of Chinook Salmon found in the bass stomach. Mean C was calculated for each reach per day from all bass examined including those with empty stomachs.

We estimated total Chinook Salmon loss to predation by multiplying daily estimates of C to abundance estimates from our previous predation research. Mean abundance during May of 2013 and 2014 equated to 1,089 Smallmouth Bass/rkm in the upper portion of our study area and mean June abundances equated to 896/rkm in the lower portion of our study area (Connor et al. 2015). Applying this to our study area equated to 21,562 bass in our reach above Pittsburg Landing, 18,949 bass in our reach below Pittsburg Landing, and 25,626 in our reach below Captain John.

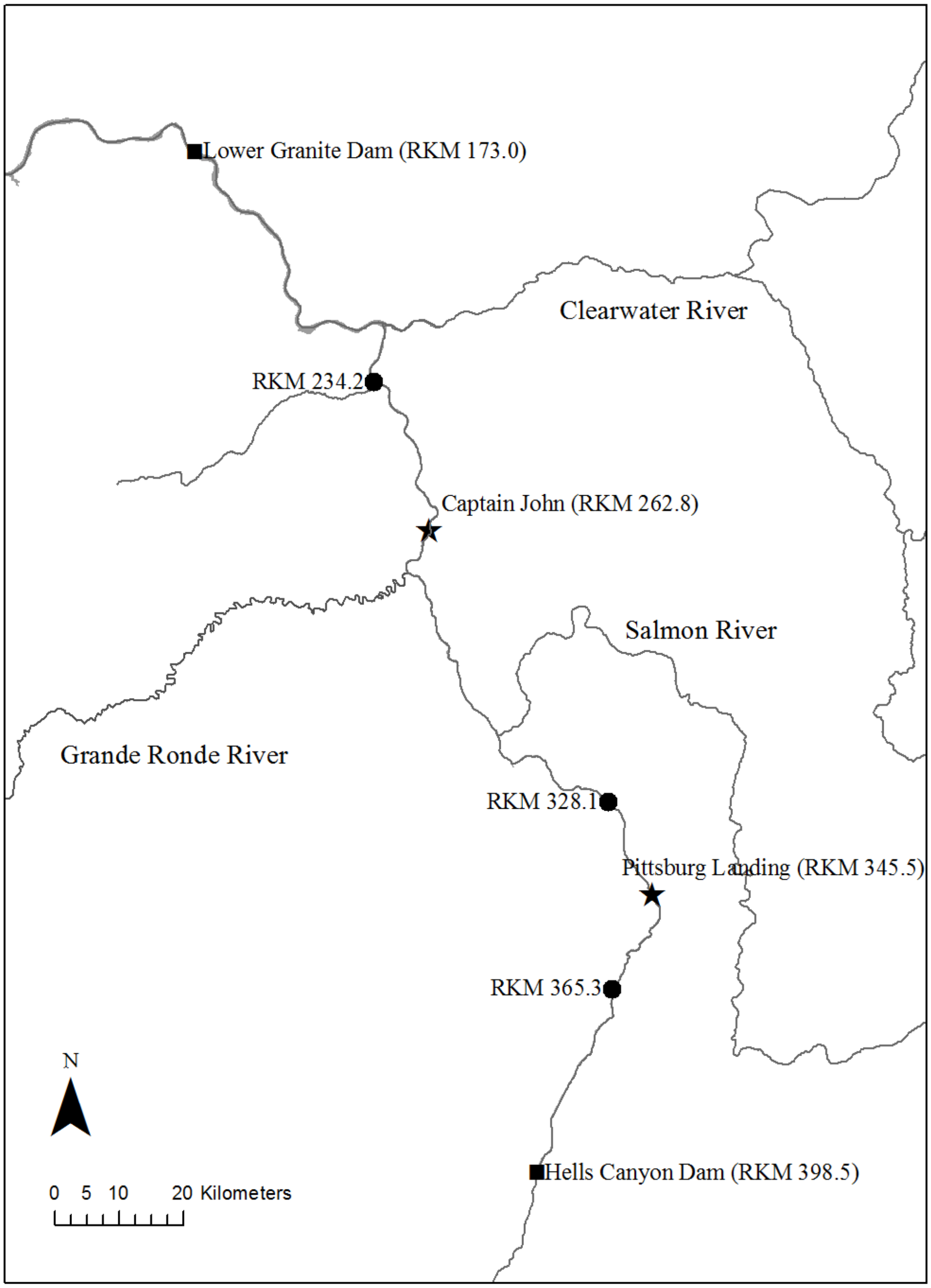

FIGURE 2. Map of Hells Canyon showing the upper (rkm 328.1–365.3) and lower (rkm 234.2–262.8) portions of our study area that were sampled in 2016.

TABLE 2. Summary of hatchery releases of fall Chinook Salmon in the Snake River during 2016.

Date	Location	Release agency	Rearing location	Number released	Number PIT tagged	Number coded-wire tagged
Releases studied						
May 16-18	Hells Canyon Dam	Idaho Power Company	Irrigon	1,041,185	2,998	247,407
May 20	Pittsburg Landing	Nez Perce Tribe	Lyons Ferry	398,086	26,052	199,287
Other releases in study area						
May 25	Captain John	Nez Perce Tribe	Lyons Ferry	509,235	25,973	200,348
May 31	Grande Ronde River	Idaho Power Company	Irrigon	429,889	3,000	200,075
June 10	Captain John	Nez Perce Tribe	Lyons Ferry	198,983	2,000	197,857

Results

Attributes of Spawning in Riverine Habitat

The total number of redds in the Snake River upper reach (estimated shallow and counted deep combined) was 1,381 in 2016 (Table 3), which ranked sixth for the 26-year period of record for that reach. The total number of redds in the Snake River lower reach was 593 in 2016 (Table 3), which ranked eleventh for the 26-year period of record for that reach. The grand total redd count for both reaches combined in 2016 was 1,974 (Table 3), which ranked eighth for the 26-year period of record for the free-flowing Snake River.

Mean total redd counts were similar between reaches during the low abundance period, whereas the mean total redd count was 1.6 times higher in the upper reach compared to the lower reach during the high abundance period (Table 3). As such, there was a large change in the percentage of the total redds counted between reaches and abundance periods with the upper reach supporting the majority of redds during the high abundance period (Table 3).

The Beverton-Holt (AIC = 351.8) provided more information about the shape of the relation between total escapement and aerial redd counts for the reaches combined compared to Ricker (AIC = 355.1) and linear (AIC = 368.9) models. The Beverton-Holt model provided the best fit (AIC = 178.6) to the data set composed of total escapement and the number of shallow spawning sites used (Linear, AIC = 229.5; Ricker, AIC = 201.9; Figure 4).

The fits to the total escapement and deepwater redd count data were similar between the Beverton-Holt (AIC = 329.8) and Ricker (AIC = 330.1) models, but better than the fit of the linear (AIC = 344.6) model (Figure 5). The Beverton-Holt model (AIC = 180.2) provided the best fit to data collected on the total escapement and the number of deepwater sites used (Linear, AIC = 206.6; Ricker, AIC = 184.3; Figure 6).

Because of the nature of the sUAS surveys, spawn timing statistics could not be calculated in 2015 using the approach applied to previous years. It can be said that spawning in the Snake River in 2015 began around the second week of October, peaked in late October to early November, and was complete by the middle of December. Based on the data collected during the 1991–2014 standardized aerial surveys, there was a 7-d difference between the mean first dates of spawning of the two abundance periods, that was partly caused by starting the aerial surveys early during several years of the high abundance period (Table 4). On average, however, relatively low proportions of the total number of redds counted within each abundance period were counted on the first date of spawning (Table 4). There was no large difference between the mean peak and last dates of spawning between abundance periods (Table 4). Thus, difference in time of spawning between abundance periods was not a large factor for changes in the attributes of juveniles described later in this report.

TABLE 3. Redd count data collected during aerial surveys (< 3 m deep; 1991–2016), sUAS surveys (< 3 m deep; 2015 and 2016) and deepwater searches (> 3 m deep; 1993–2016) conducted along the Snake River upper and lower reaches, 1991–2016. The mean (± SD) total counts and mean (± SE) percentages by reach are also given by abundance period (Low, 1991–1998; High, 1999–2016).

	Upper reach			Lower reach			Grand	Percent by reach	
Year	Aerial	Deep	Total	Aerial	Deep	Total	total	Upper	Lower
1991	27		27	24	5	29	56	48.2	51.8
1992	16		16	31	0	31	47	34.0	66.0
1993	14	5	19	46	62	108	127	15.0	85.0
1994	29	6	35	22	8	30	65	53.8	46.2
1995	28	5	33	13	19	32	65	50.8	49.2
1996	49	7	56	22	26	48	104	53.8	46.2
1997	20	4	24	29	5	34	58	41.4	58.6
1998	109	28	137	26	22	48	185	74.1	25.9
1999	225	67	292	48	33	81	373	78.3	21.7
2000	186	42	228	74[a]	49	123	351	65.0	35.0
2001	301	87	388	234	86	320	708	54.8	45.2
2002	533	114	647	345	120	465	1,112	58.2	41.8
2003	675	165	840	455	229	684	1,524	55.1	44.9
2004	685	279	964	533	210	743	1,707	56.5	43.5
2005	662	203	865	380	195	575	1,440	60.1	39.9
2006	452	147	599	244	181	425	1,024	58.5	41.5
2007	482	241	723	232	162	394	1,117	64.7	35.3
2008	761	368	1,129	472	218	690	1,819	62.1	37.9
2009	948	379	1,327	563	205	768	2,095	63.3	36.7
2010	1,110	641	1,751	840	375	1,215	2,966	59.0	41.0
2011	874	521	1,395	1,075	344	1,419	2,814	49.6	50.4
2012	1,100[b]	274	1,374	594[c]	142	736	2,110	65.1	34.9
2013	1,338[d]	332	1,670	1,209[e]	264	1,473	3,143	53.1	46.9
2014	1,949[f]	583	2,532	1,029[g]	259	1,288	3,820	66.3	33.7
2015	1,738[h]	589	2,327	591[i]	237	828	3,155	73.8	26.2
2016	1,043[j]	338	1,381	445[k]	148	593	1,974	70.0	30.0
Low			43 ± 37	45 ± 25			88 ± 45	46.4 ± 16.1	53.6 ± 16.1
High			1,135 ± 634	712 ± 399			1,847 ± 982	61.9 ± 1.7	38.1 ± 1.7

[a]Groves et al. (2013) did not report 5 redds counted at rkm 238.1; [b]1,016/0.924 based on observed helicopter counts divided by counts made with a UAS;[c]396/0.667; [d]1,220/0.912; [e]851/0.704;[f]1,129/0.579;[g]837/0.813;[h]± 670 redds;[i]± 222 redds;[j]± 96redds; [k]± 138redds.

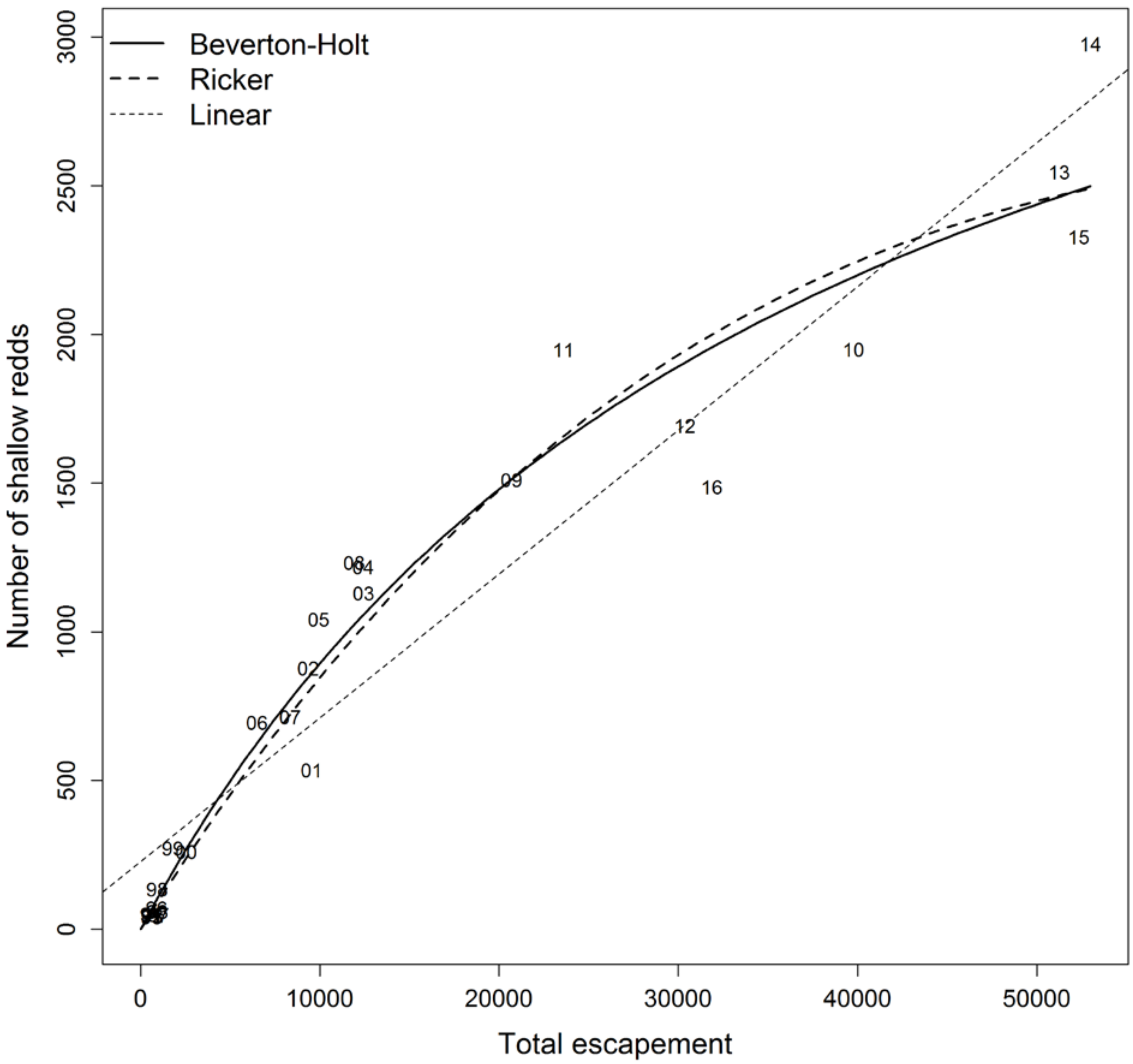

FIGURE 3. The relation between estimated annual total escapement (Table 1) and the annual number of redds counted (1992–2010) counted and adjusted for inaccuracy (2012–2014), and estimated (2015 and 2016) based on manned helicopter (1992–2014) and unmanned aircraft flights over shallow (< 3 m deep) spawning sites along the Snake River upper and lower reaches combined (Table 3), 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 0.11293±0.01244 and beta = 0.000026±0.000006; and alpha = 0.09704±0.00815 and beta = 0.000014±0.0000021.

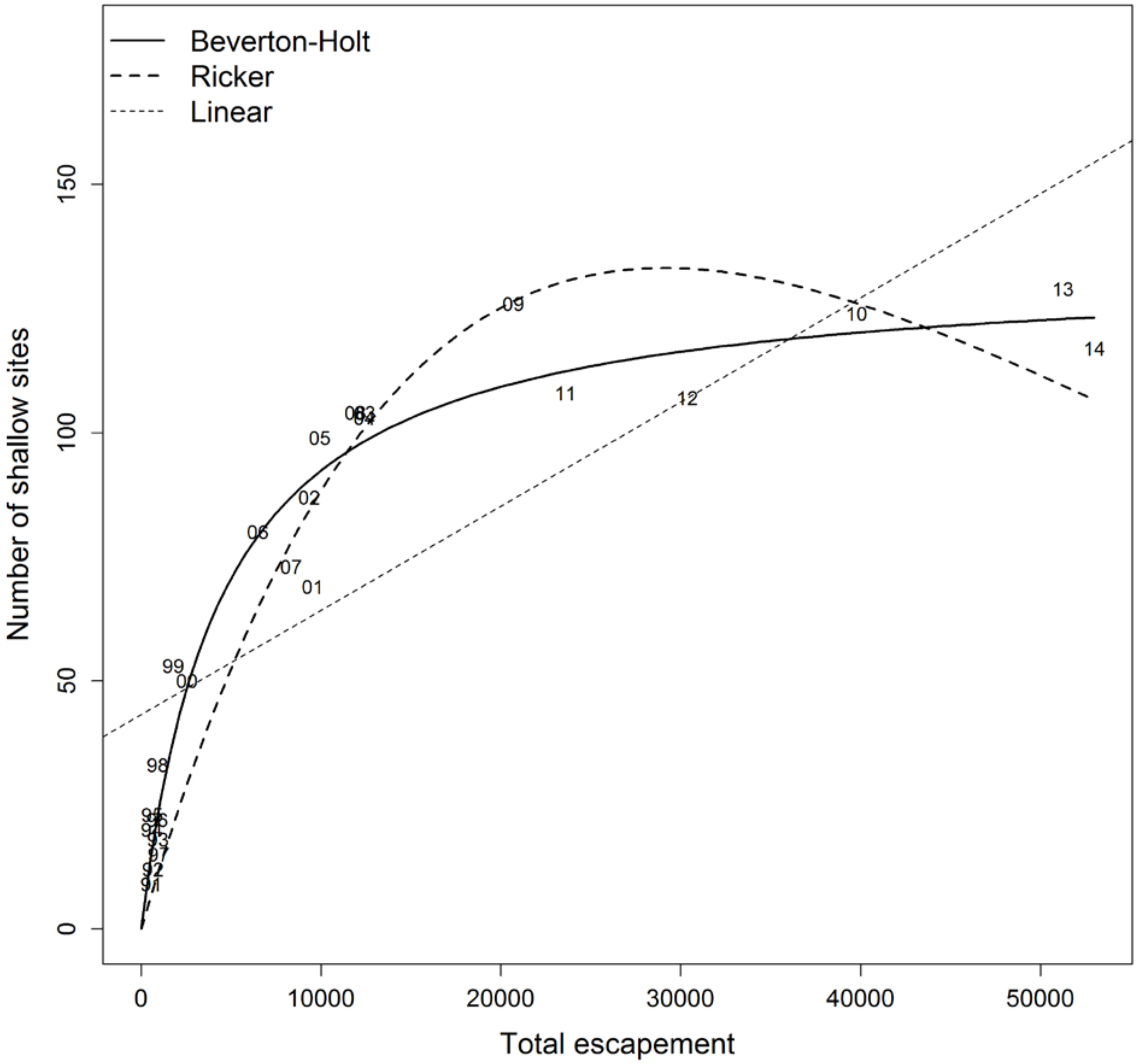

FIGURE 4. The relation between estimated annual total escapement (Table 1) and the annual number of shallow (< 3 m deep) sites at which redds were counted from the air in the Snake River upper and lower reaches combined, 1991–2014. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt model (the best fit) were alpha = 0.029912±0.0036 and beta = 0.000224±0.00003.

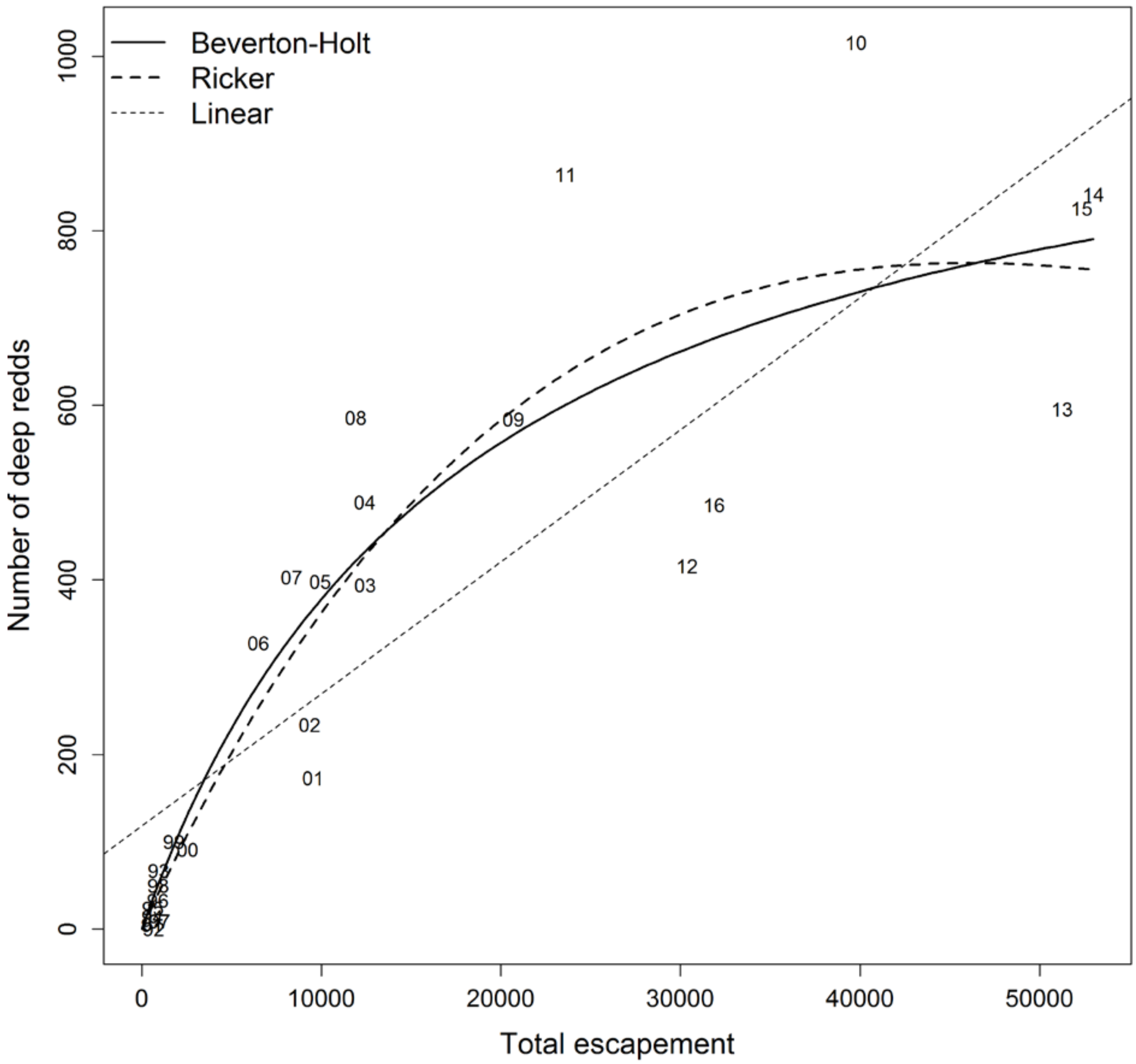

FIGURE 5. The relation between estimated annual total escapement (Table 1) and the annual number of redds counted at sites > 3 m deep with underwater video along the Snake River upper and lower reaches combined (Table 3), 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 0.058803±0.0134256 and beta = 0.0000555±0.0000203; and alpha = 0.0450296±0.00589and beta = 0.000022±0.000004.

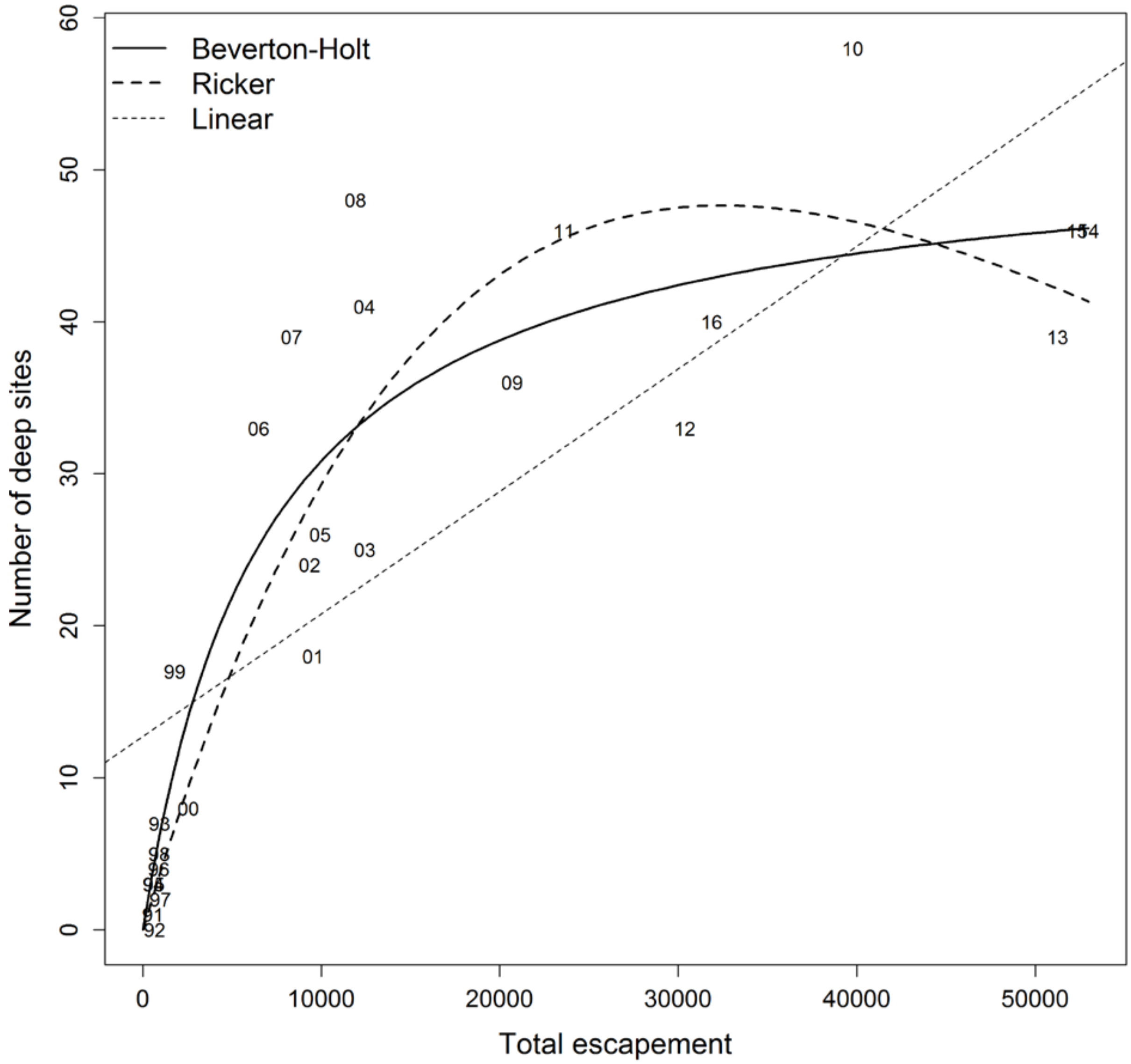

FIGURE 6. The relation between estimated annual total escapement (Table 1) and the annual number of deep (> 3 m deep) sites at which redds were counted with underwater video in the Snake River upper and lower reaches combined, 1991–2016. The numbers are the last two digits of the calendar year. The parameters (± SE) for the Beverton-Holt model (the best fit) were alpha = 0.00755±0.0018 and beta = 0.000145±0.000046.

TABLE 4. Information from the annual aerial surveys made to count redds in shallow water (< 3 m) along the Snake River upper and lower reaches including the number of flights made, the spawning date represented by the first flight (flight date minus 7 d; Earliest), and the first, peak, and last spawning dates (flight dates minus 7 d) observed including the cumulative percentages (%) of eventual total redd counts made on those dates. The means (± SD) of the annual abundance period (Low, 1991–1998; High, 1999–2014) first, peak and last dates redds were counted are also given. sUAS survey data collected in 2015 and 2016 were not comparable to early years with regards to evaluating time of spawning.

	Flights		First		Peak		Last	
Year	*N*	Earliest	Date	%	Date	%	Date	%
1991	9	10/07	10/21	2.2	11/11	26.2	12/02	4.8
1992	8	10/09	10/29	8.5	11/16	38.3	12/05	6.4
1993	8	10/18	10/18	1.7	10/25	30.0	12/06	3.3
1994	8	10/17	10/17	2.0	10/31	35.3	11/28	2.0
1995	7	10/16	10/16	9.8	10/30	43.9	11/20	2.4
1996[a]	7	10/14	10/14	2.8	10/21	36.6		
1997	8	10/13	10/13	2.0	10/20	32.7	11/24	2.0
1998	8	10/12	10/19	20.7	10/26	31.9	11/23	3.7
1999	9	10/04	10/11	1.8	11/04	43.6	11/30	0.4
2000	9	10/02	10/02	0.4	10/23	31.4	11/21	2.0
2001	10	10/02	10/02	1.7	11/06	39.8	12/03	1.5
2002	7	10/14	10/14	3.4	10/28	33.1	11/25	0.3
2003	7	10/13	10/13	0.4	11/03	37.4	12/01	0.5
2004	8	10/11	10/11	0.1	11/01	40.5	11/29	1.1
2005	9	10/11	10/11	0.8	10/31	32.5	12/05	0.1
2006	6	10/16	10/16	5.0	10/23	39.3	11/27	1.3
2007	8	10/08	10/08	0.1	10/30	39.5	11/26	6.0
2008	8	10/13	10/13	1.9	10/27	46.0	12/01	0.5
2009	8	10/12	10/12	1.2	10/26	46.4	11/30	0.9
2010	4	10/18	10/18	3.2	11/02	51.5	11/28	6.9
2011	5	10/17	10/17	1.7	10/31	47.5	11/28	6.7
2012	4	10/15	10/15	2.4	10/29	68.5	11/26	1.8
2013	4	10/14	10/14	5.0	10/28	75.6	11/27	0.8
2014	4	10/13	10/13	1.9	10/27	53.8	11/24	2.4
Low		10/14±4	10/19±5	6.2±6.3	10/31±9	34.1±5.6	11/28±6	3.1±1.5
High		10/12±5	10/12±4	1.9±1.5	10/30±4	45.4±11.9	11/28±3	2.2±2.2

[a]Counts were prohibited by turbidity during the last three flights.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat

Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River upper reach during 2016 was 37.1 ± 21.5 fish per seine haul (Table 5), which ranked seventh for the 22-year period of record for the upper reach. Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River lower reach during 2016 was 19.9 ± 4.8 fish per seine haul (Table 5), which ranked ninth for the 25-year period of record for the lower reach. Mean (± 95% C.L.) annual CPUE for natural-origin Snake River fall Chinook Salmon subyearlings along the Snake River upper and lower reaches combined during 2015 was 25.3 ± 7.5 fish per seine haul (Table 5), which ranked sixth for the 25-year period of record. Mean inter-annual CPUE was lower during the low abundance period compared to the high abundance period for the upper reach, lower reach, and the combined reaches (Table 5).

The Ricker (AIC = 193.6) and Beverton-Holt (AIC =195.5) models provided similar amounts of information about the shape of the relation between estimated total escapement and mean annual CPUE calculated jointly between Snake River reaches, but more information about that relation than was provided by the linear model (AIC = 207.7; Figure 7).

TABLE 5. Mean (± 95% C.I.) CPUE (fish per seine haul) for natural-origin fall Chinook Salmon subyearlings along the Snake River upper and lower reaches, 1992–2016. The start and end dates for beach seining, the total number of station visits (*N*), and the grand means (± SE) of the abundance period (Low, 1992–1999; High, 2000–2016) annual means are also given.

	Upper reach				Lower reach				Combined			
Year	*N*	Start	End	CPUE	*N*	Start	End	CPUE	*N*	Start	End	CPUE
1992					173	04/01	06/11	3.5±0.9	173	04/01	06/11	3.5±0.9
1993					247	04/06	07/20	1.5±0.3	247	04/06	07/20	1.5±0.3
1994					249	04/06	07/13	6.0±1.8	249	04/06	07/13	6.0±1.8
1995	70	04/07	06/29	8.2±2.7	199	04/05	07/06	3.3±0.8	269	04/05	07/06	4.6±0.9
1996	54	04/18	07/11	0.8±0.3	145	04/16	07/17	2.2±0.5	199	04/16	07/17	1.8±0.4
1997	71	04/24	07/03	0.6±0.3	167	04/22	07/16	2.5±0.7	238	04/22	07/16	1.9±0.5
1998	73	04/15	07/06	4.7±1.8	149	04/14	07/08	5.2±1.3	222	04/14	07/08	5.0±1.1
1999	81	04/08	07/09	8.8±3.4	171	04/06	07/15	4.2±0.9	252	04/06	07/15	5.7±1.2
2000	41	04/06	06/29	31.4±24.2	98	04/04	07/06	17.5±5.0	139	04/04	07/06	21.6±7.8
2001	49	04/06	06/21	11.3±8.2	140	04/04	07/03	11.1±3.0	189	04/04	07/03	11.2±3.1
2002	56	04/04	07/11	29.6±16.6	160	04/02	07/17	7.8±1.8	216	04/02	07/17	13.5±4.6
2003	52	03/27	06/26	40.1±16.6	146	03/25	07/02	19.7±4.8	198	03/25	07/02	25.1±5.7
2004	55	03/25	06/24	90.9±49.2	150	03/23	06/30	41.8±11.3	205	03/23	06/30	54.9±15.6
2005	60	03/31	06/30	88.7±45.9	199	03/29	07/12	39.7±8.6	259	03/29	07/12	51.0±12.6
2006	78	03/31	07/06	7.0±3.7	216	03/29	07/11	5.8±1.2	294	03/29	07/11	6.1±1.3
2007	68	03/29	07/05	65.2±38.5	144	03/27	07/03	23.1±5.4	212	03/27	07/05	36.6±13.0
2008	92	03/27	07/17	34.7±14.4	169	03/25	07/16	10.2±2.2	261	03/25	07/17	18.8±5.4
2009	79	03/26	06/25	25.6±13.7	155	03/23	07/07	14.5±5.1	234	03/23	07/07	18.3±5.7
2010	86	03/25	07/22	20.6±8.0	187	03/23	07/27	26.7±5.3	273	03/23	07/27	24.8±4.4
2011	79	03/31	07/14	20.4±8.4	161	03/29	07/20	13.8±3.7	240	03/29	07/20	16.0±3.7
2012	85	03/28	07/11	49.0±22.4	178	03/27	08/01	21.2±4.5	263	03/27	08/01	30.2±7.9
2013	68	03/28	06/27	65.0±46.2	145	03/26	07/03	28.8±6.5	213	03/26	07/03	40.4±15.3
2014	60	03/27	06/19	24.9±13.0	133	03/25	07/01	20.5±4.4	193	03/25	07/01	21.9±5.0
2015	54	03/26	06/11	19.5±9.7	125	03/24	06/24	27.5±5.1	179	03/24	06/24	25.1±5.0

TABLE 5. (Extended)

	Upper reach				Lower reach				Combined			
Year	*N*	Start	End	CPUE	*N*	Start	End	CPUE	*N*	Start	End	CPUE
2016	65	03/24	06/16	37.1±21.5	141	03/22	06/28	19.9±4.8	206	03/22	06/28	25.3±7.5
Low				4.6 ±1.7				3.6 ±0.5				3.8±0.6
High				38.9±6.3				20.6±2.5				25.9±3.3

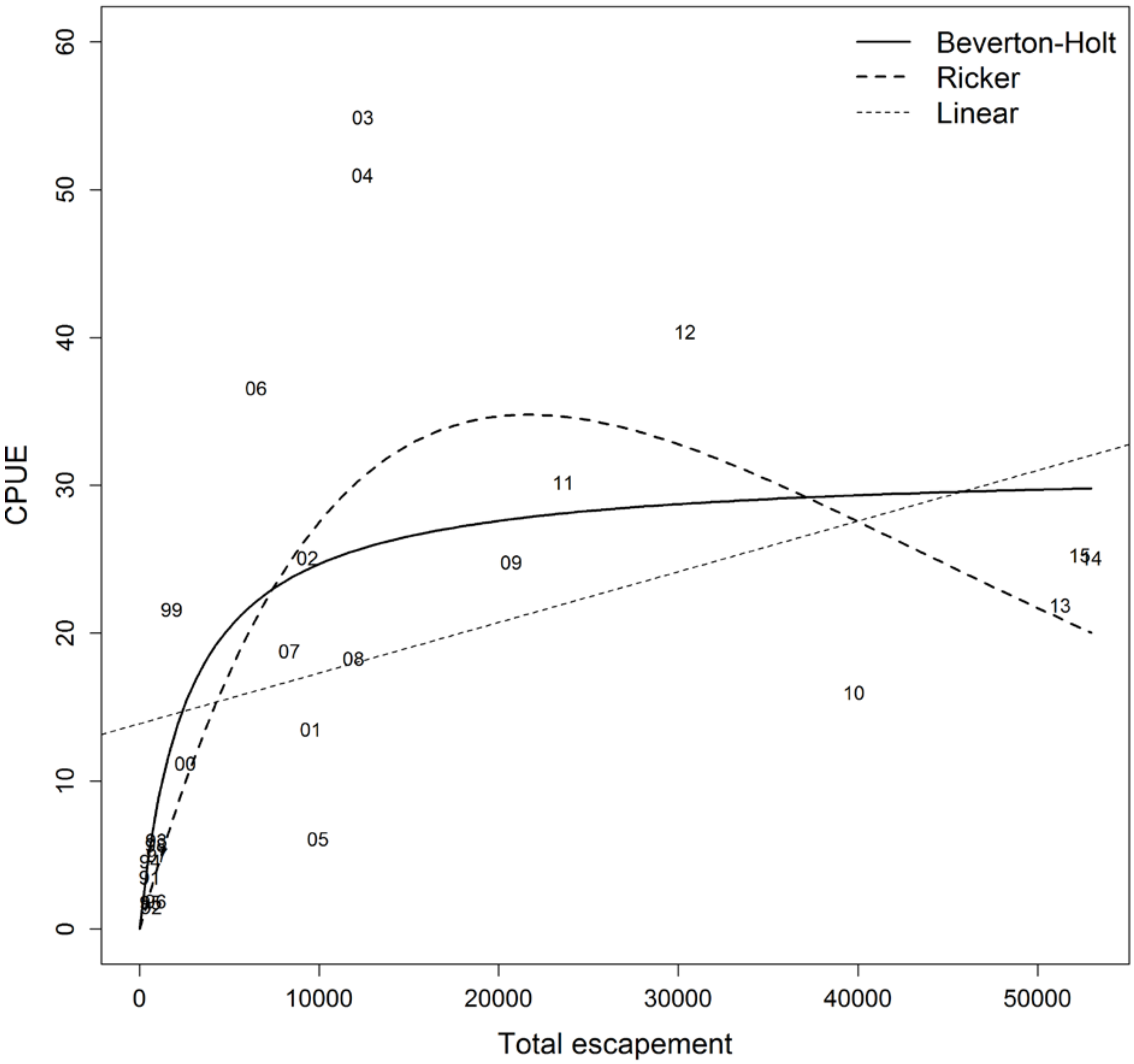

FIGURE 7. The relation between estimated annual total escapement (Table 1; 1991–2016) and annual mean CPUE of natural-origin Fall Chinook Salmon subyearlings along the Snake River upper and lower reaches combined (Table 5; 1992–2016). The numbers are the last two digits of the brood year (e.g., 15 = CPUE for juveniles in 2016 produced by spawning in 2015). The parameters (± SE) for the Ricker model (the best fit) were alpha = 0.011673±0.006642 and beta = 0.000373± 0.000252.

Of the life stages evaluated, timing of fry presence has been the most stable across the years 1992–2016 with emergence timing generally being earlier in the relatively warmer Snake River upper reach than in the relatively cooler Snake River lower reach in most years (Table 6). The median date of parr presence along the shorelines is an indicator of when the number of parr rearing in riverine habitat became less than the number of parr that had begun downstream dispersal into Lower Granite Reservoir. The median date of parr presence along both Snake River reaches generally became earlier as density along the shorelines increased as can be seen by examining the means of the median dates of parr presence for the periods of low (1992–1999) and high (2000–2016) abundance (Table 7).

Grand mean (± SE) wet weight of natural-origin parr in the Snake River upper reach declined from 4.7 ± 0.4 g during the low abundance period to 2.4 ±0.2 g during the high abundance period, and a similar decline in wet weight of parr was observed in the Snake River lower reach (Table 8). The decline in wet weight of parr was accompanied by a decline in growth in weight in both reaches, but especially in the upper reach (Table 8) where rearing habitat availability and connectivity are relatively low and apparent abundance high (i.e., CPUE in Table 5).

TABLE 6. Timing of natural-origin fry presence (Sunday's day of year; January 1 = 1) along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) medians are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					356	117	89	145
1993					199	136	94	171
1994					441	135	93	156
1995	117	113	92	141	257	120	92	155
1996	14	119	105	126	268	126	105	175
1997	1	110	110	110	114	124	110	180
1998	101	109	102	130	322	116	102	165
1999	97	122	94	143	278	122	94	178
2000	683	100	93	135	415	100	93	156
2001	552	119	91	140	1,268	126	91	154
2002	2,289	111	90	153	965	125	90	167
2003	962	103	82	145	1,726	110	82	173
2004	6,123	109	81	144	4,952	123	81	158
2005	5,462	107	86	135	3,786	107	86	156
2006	75	106	85	141	588	134	85	162
2007	4,311	112	84	154	1,771	119	84	154
2008	1,628	118	90	146	1,231	118	83	167
2009	811	109	81	137	1,017	116	81	165
2010	1,572	115	80	157	4,393	115	80	171
2011	1,778	128	86	163	1,408	128	86	191
2012	3,782	120	85	169	2,207	127	85	162
2013	7,874	111	83	146	4,033	118	83	167
2014	1,174	107	82	159	2,755	117	82	166
2015[a]	1,349	102	81	144	5,678	102	81	158
2016	866	101	80	129	2,164	108	80	178
Low		115 ± 3				125 ± 3		
High		110 ± 2				118 ± 2		

[a]During week 4 of the seining season a new seine was used. That seine was found to have a mesh size of 1/4 inch instead of the standard 3/16 inch. Catch was adjusted by dividing the week 4 catch by 0.75. The seine used after before and after week 4 had 3/16 inch mesh.

TABLE 7. Timing of natural-origin parr presence (Sunday's day of year; January 1 = 1) along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) medians are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					1,765	138	89	159
1993					2,166	157	101	199
1994					4,348	149	93	191
1995	985	148	99	169	1,408	155	92	183
1996	118	133	105	168	756	147	105	196
1997	119	145	110	166	938	159	110	194
1998	1,078	137	102	186	2,512	151	102	186
1999	1,493	143	101	178	1,647	157	94	192
2000	1,064	114	93	163	1,578	135	93	177
2001	794	123	91	161	3,076	140	91	175
2002	3,013	125	97	181	3,620	146	90	188
2003	4,523	124	82	173	6,821	131	82	180
2004	6,310	123	88	172	11,225	137	88	179
2005	8,119	121	86	170	16,803	135	86	184
2006	1,344	134	85	176	2,658	134	99	176
2007	7,226	119	91	182	7,500	133	84	182
2008	6,610	139	97	195	3,357	146	83	195
2009	3,876	130	88	165	4,706	137	95	186
2010	2,502	129	87	199	9,193	143	87	199
2011	2,237	135	93	184	2,759	149	86	191
2012	4,769	134	106	176	5,791	148	99	190
2013	4,503	125	104	174	7,171	139	83	174
2014	2,484	131	82	159	3,785	138	89	180
2015[a]	2,221	109	81	151	7,050	137	81	172
4,210	115	94	157		4,980	122	87	178
Low		141 ± 3				152 ± 2		
High		125 ± 2				138 ± 2		

[a]During week 4 of the seining season a new seine was used. That seine was found to have a mesh size of 1/4 inch instead of the standard 3/16 inch. Catch was adjusted by dividing the week 4 catch by 0.75. The seine used after before and after week 4 had 3/16 inch mesh.

TABLE 8. Mean (± SD) wet weights (0.1 g) and absolute growth rates (0.01 g/d) of natural-origin parr rearing along the Snake River upper and lower reaches, 1992–2016. The means (± SE) of the annual abundance period (Low, 1992–1999; High, 2000–2016) grand means are also given.

	Upper reach				Lower reach			
Year	*N*	Weight	*N*	Growth	*N*	Weight	*N*	Growth
1992					1,128	4.2±2.5	36	0.17±0.11
1993					1,901	4.2±3.1	161	0.13±0.13
1994					3,712	3.8±2.8	238	0.19±0.17
1995	606	4.4±3.2	29	0.28±0.13	888	4.0±3.3	35	0.21±0.11
1996	112	3.8±2.5	17	0.26±0.13	714	4.2±3.4	49	0.20±0.14
1997	114	6.1±2.7	20	0.34±0.10	922	4.7±3.1	78	0.20±0.11
1998	981	4.7±3.0	89	0.25±0.11	2,145	4.0±2.8	86	0.18±0.09
1999	1,489	4.3±3.0	169	0.30±0.13	1,642	3.7±2.8	92	0.24±0.12
2000	932	4.2±3.9	61	0.38±0.11	1,553	3.7±3.1	45	0.20±0.10
2001	724	2.0±1.3	11	0.22±0.03	2,981	2.6±1.8	120	0.18±0.07
2002	3,005	2.1±1.6	169	0.24±0.09	3,620	3.1±2.3	185	0.17±0.07
2003	4,480	2.5±1.5	359	0.18±0.10	6,821	2.5±1.5	186	0.14±0.06
2004	4,031	2.2±1.4	150	0.18±0.10	10,373	2.6±1.7	422	0.17±0.06
2005	7,739	2.1±1.2	218	0.20±0.08	15,995	2.4±1.6	353	0.18±0.07
2006	1,138	3.4±1.8	46	0.22±0.12	2,648	2.7±1.9	52	0.20±0.10
2007	7,223	1.7±1.0	128	0.21±0.08	7,318	2.9±1.9	425	0.19±0.07
2008	5,570	3.2±2.3	770	0.17±0.10	3,272	2.9±2.1	168	0.15±0.08
2009	3,348	2.2±1.0	121	0.09±0.03	4,478	2.5±1.6	265	0.10±0.07
2010	1,754	2.3±1.6	184	0.11±0.07	8,080	2.4±1.7	759	0.09±0.07
2011	1,544	2.1±1.6	139	0.15±0.09	2,454	2.5±1.7	149	0.13±0.08
2012	3,772	1.9±1.1	133	0.12±0.05	4,973	2.9±1.9	512	0.14±0.07
2013	2,092	2.0±1.1	149	0.12±0.10	5,534	2.7±1.9	436	0.11±0.06
2014	1,253	2.4±1.6	96	0.23±0.10	2,669	2.5±1.8	165	0.12±0.07
2015	1,354	1.7±0.7	27	0.09±0.04	4,618	2.6±1.8	291	0.11±0.06
2016	2,217	1.7±0.8	102	0.09±0.05[a]	3,614	2.3±1.6	225	0.09±0.05
Low		4.7±0.4		0.29±0.02		4.1±0.1		0.19±0.01
High		2.4±0.2		0.18±0.02		2.7±0.1		0.15±0.01

[a]Does not include fish< 50-mm FL experimentally tagged with 8-mm tags.

Attributes of Natural-Origin Juveniles Emigrating through Lower Granite Reservoir

The estimate of passage abundance for the basin-wide population of natural-origin, subyearling smolts at Lower Granite Dam in 2015 should be viewed with caution (i.e., it may be an overestimate; Table 9) because the low flow levels and high spill percentages at Lower Granite Dam in 2015 were not represented in the process of modeling collection probability at the dam. As such, the estimates are extrapolations and are not considered to be reliable at this time. Estimated passage abundance in 2016, when environmental conditions were in the range used for modeling fitting, is given in (Table 9).

The Beverton-Holt model (AIC = 697.4) provided more information about the shape of the relation between total escapement and estimated passage abundance of the basin-wide population of subyearling smolts compared to the Ricker (AIC = 700.6) and linear (AIC = 703.1) models (Figure 8).

Passage of PIT-tagged natural-origin subyearling smolts at Lower Granite Dam was the second earliest on record for fish from both the Snake River upper and lower reaches (Table 10). The median (± SE) date of passage of PIT-tagged, natural-origin subyearling smolts from the Snake River upper reach was later during the low abundance period compared to the high abundance period (Table 10; 8-d difference). The same pattern was observed for smolts that had been PIT tagged as parr in the lower reach, but the difference was greater than was observed for smolts from the upper reach (Table 10; a 13-d difference). In addition to the change in dispersal timing of parr from riverine habitat observed between abundance periods (Table 7), changes in downstream movement rate also contributed to the general shift in passage timing at Lower

Granite Dam between abundance periods. Smolts that had been PIT-tagged as parr rearing along the Snake River upper reach moved downstream slower on average during the low abundance period compared to the high abundance period (Table 11; 1.5-km/d difference). A similar shift in rate of downstream movement was observed for smolts that had been PIT tagged as parr in the lower reach, but the difference was not as great as observed for smolts from the upper reach (Table 11; a 0.6-km/d difference).

TABLE 9. Estimated (N^; 95% lower C.L.; 95% upper C.L.) passage abundance of natural-origin, subyearling fall Chinook Salmon smolts from the basin-wide population at Lower Granite Dam, 1992–2016. The means (± SE) of the annual estimates for the abundance periods (Low, 1992–1999; High, 2000–2016) are also given. The method is under development. Passage abundance was not estimated from November of year t through the third week of March year $t + 1$ when the Smolt Monitoring Program was not in operation and juvenile fish bypass system was dewatered.

Year	N^	95% Lower C.L.	95% Upper C.L.
1992	13,672	12,236	21,002
1993	15,222	14,593	19,387
1994	15,895	15,165	22,403
1995	82,797	81,367	109,757
1996	36,752	35,725	48,805
1997	298,553	274,731	461,608
1998	130,765	124,893	192,413
1999	303,808	280,765	438,467
2000	585,424	471,100	1,172,425
2001	446,497	406,453	633,339
2002	255,237	216,226	405,064
2003	683,169	663,222	873,430
2004	1,177,956	1,118,431	1,514,057
2005	558,317	481,085	910,945
2006	268,364	257,713	394,865
2007	197,907	120,629	370,693
2008	429,650	398,996	644,548
2009	406,498	385,639	552,105
2010	849,839	809,427	1,235,190
2011	423,060	419,057	554,740
2012	584,172	559,331	825,038
2013	654,498	638,684	942,771
2014	486,115	448,715	709,970
2015	1,357,482	1,217,318	2,240,308
2016	530,458	248,436	2,662,701
Low	112,183 ± 43,656		
High	582,038 ± 74,422		

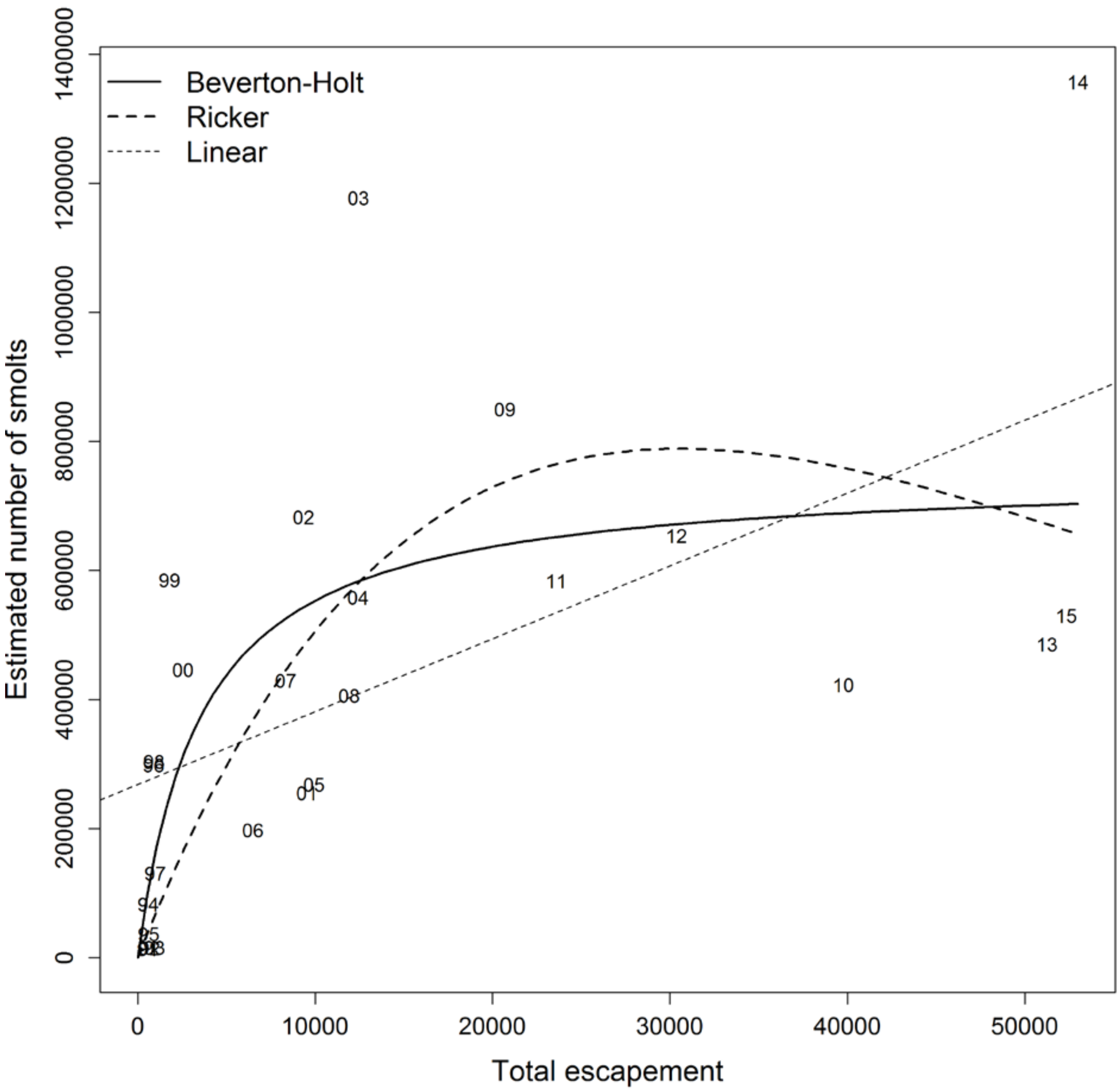

FIGURE 8. The relation between estimated annual total escapement (Table 1) and the estimated annual passage abundance of natural-origin, subyearling smolts from the basin-wide population at Lower Granite Dam (Table 9), 1992–2016. Passage abundance was not estimated from November of year *t* through the third week of March year *t + 1*. The numbers are the last two digits of the brood year (e.g., 15 = smolt abundance in 2016 produced by spawning in 2015). The parameters (± SE) for the Beverton-Holt and Ricker models (of similar fit) were: alpha = 1210.38±119.23and beta = 0.00020±0.0002; and alpha = 70.21±15.3869 and beta 0.000033±0.000007.

TABLE 10. Median dates of passage at Lower Granite Dam for natural-origin, subyearling smolts that were PIT tagged along the Snake River upper and lower reaches, 1992–2016. Estimated annual passage abundance (*N*^) of the PIT-tagged fish, and the means (± SE) of the abundance period medians (Low, 1992–1999; High, 2000–2016) are also given.

	Upper reach		Lower reach	
Year	*N*^	Median	*N*^	Median
1992			85	06/22
1993			483	07/22
1994			450	07/17
1995	401	07/18	490	08/02
1996	47	06/29	256	07/19
1997	55	06/20	192	07/16
1998	349	07/07	739	07/11
1999	712	06/26	556	07/27
2000	203	06/19	532	07/02
2001	20	07/12	374	07/07
2002	310	07/01	1,156	07/04
2003	642	06/24	1,694	06/27
2004	337	06/24	2,907	06/24
2005	1,059	06/20	2,938	06/24
2006	335	06/15	411	06/27
2007	238	06/18	1,243	06/27
2008	1,382	07/01	957	07/17
2009	405	06/22	1,023	07/03
2010	263	07/02	2,014	07/12
2011	402	06/17	589	07/12
2012	674	06/25	1,555	07/09
2013	179	07/01	903	07/04
2014	249	07/02	971	07/15
2015	83	05/28	2,473	06/23
2016	283[a]	06/12	1,994	06/16
Low	07/07 (4 d)		07/17 (4 d)	
High	06/24 (2 d)		07/03 (2 d)	

[a]Does not include fish experimentally tagged with 8-mm tags.

TABLE 11. Median, minimum, and maximum rates of downstream movement (km/d) for natural-origin parr that were PIT tagged along the Snake River upper and lower reaches and subsequently detected at Lower Granite Dam as subyearling smolts, 1992–2016. The means (± SE) of the abundance period medians (Low, 1992–1999; High, 2000–2016) are also given.

	Upper reach				Lower reach			
Year	*N*	Median	Min	Max	*N*	Median	Min	Max
1992					39	3.3	1.1	18.8
1993					233	1.5	0.7	19.7
1994					194	1.5	0.4	13.8
1995	203	3.4	1.1	44.8	239	1.5	0.4	20.3
1996	19	3.7	2.1	43.8	126	1.5	0.4	19.3
1997	24	5.3	1.7	70.5	99	2.5	0.5	21.6
1998	173	3.4	1.6	20.1	380	2.0	0.4	28.3
1999	326	4.7	1.0	35.0	280	2.1	0.4	19.5
2000	72	5.3	2.6	19.9	257	1.9	0.4	13.6
2001	10	3.6	1.3	4.5	185	1.8	0.5	15.0
2002	95	4.6	2.5	25.0	395	2.3	0.8	19.3
2003	304	5.3	2.3	43.8	814	2.5	0.5	29.5
2004	186	5.0	2.9	21.9	1,597	2.6	0.7	21.3
2005	314	6.0	2.9	26.3	672	3.4	1.0	19.7
2006	96	8.0	3.5	58.3	103	3.0	0.6	45.0
2007	37	5.3	3.8	13.5	177	2.8	0.4	15.6
2008	359	5.3	1.2	87.5	211	2.1	0.5	57.0
2009	145	4.9	2.2	29.8	319	2.5	0.4	42.5
2010	72	6.7	2.8	61.7	475	2.9	0.3	58.0
2011	167	7.6	2.4	64.3	231	3.4	0.5	59.0
2012	163	5.0	2.2	59.7	303	2.6	0.3	42.5
2013	33	4.1	2.6	29.2	143	2.3	0.4	21.8
2014	61	5.4	2.1	46.3	198	2.2	0.5	27.3
2015	17	5.0	3.1	20.6	296	2.8	1.1	18.0
2016	45[a]	8.1	2.7	25.6	379	2.6	1.1	42.5
Low		4.1 ± 0.4				2.0 ± 0.2		
High		5.6 ± 0.3				2.6 ± 0.1		

[a]Does not include fish experimentally tagged with 8-mm tags.

Seasonal Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat

The consumption rate (*C*) of Chinook Salmon by Smallmouth Bass increased following all four hatchery releases. Before the first release at Hells Canyon Dam, daily estimates of *C* ranged from 0 to 0.052 fish/bass/d, but increased to 0.338 fish/bass/d by May 17 (the day after the release) in our study section above Pittsburg Landing (Figure 9; actual). Pre-release consumption below Pittsburg Landing ranged from 0.012–0.355 fish/bass/d. No sampling was conducted on May 18 but we assume that *C* decreased to a pre-release level following a similar pattern to the release on May 18 (Figure 9; hypothetical). Following the second release of hatchery fish on May 18, *C* increased to 0.585 fish/bass/d above Pittsburg Landing and to 0.653 fish/bass/d below by May 19. On May 20, *C* decreased to near pre-release levels above Pittsburg Landing and remained low on May 21 as well. However, below Pittsburg Landing, *C* increased to a high of 1.604 fish/bass/d on the day of the release (May 20) at Pittsburg Landing. Consumption rate decreased slightly to 1.322 fish/bass/d on May 21 before dropping below pre-release levels on May 22.

We used the above information to estimate the loss of Chinook Salmon to Smallmouth Bass in the upper portion of our study area. We conclude that hatchery releases from Hells Canyon Dam produce a sharp, 1-d increase in Smallmouth Bass consumption while releases from Pittsburg Landing produce a 2-d increase in consumption. We assume that consumption rates in the reach below Pittsburg Landing followed a similar temporal pattern as in the reach upstream resulting in returns to pre-release levels on May 18 following the dam release on May 16 (Figure 9; hypothetical). Using this assumption, we estimate 110,177 subyearlings were consumed during 2016 in the upper portion of our Hells Canyon study area of which ~98,000 were hatchery origin based on sizes of natural and hatchery origin subyearlings consumed (Figure 10). We estimate that 38,669 hatchery subyearlings released at Hells Canyon Dam were consumed by Smallmouth Bass both above and below Pittsburg Landing. This equates to a consumption of 1,039 subyearlings/rkm in our study area and 3.65% of the total release. By individual release, we estimate 13,684 (368 subyearlings/rkm) subyearlings were consumed during the May 16 release and 24,986 (672 subyearlings/rkm) subyearlings were consumed during the May 18 release. For the Pittsburg release, we estimate 59,123 (14.85% of the release) subyearlings were consumed, which equates to 3,398 subyearlings/rkm.

Smallmouth Bass consumption rate was much lower during the June release of subyearlings from the Captain John acclimation facility. Consumption increased from a pre-release rate of 0.030 fish/bass/d on June 10 to a peak of 0.183 fish/bass/d on June 11and declined gradually thereafter to pre-release levels of 0.022 fish/bass/d by June 15 (Figure 11). We estimate 13,728 Chinook Salmon were consumed by Smallmouth Bass between Captain John and the head of Lower Granite Reservoir. Assuming bass mainly consumed hatchery fish following the release from the Captain John, we estimate 12,405 hatchery subyearlings (0.06% of the release) were lost to predation in our lower study area.

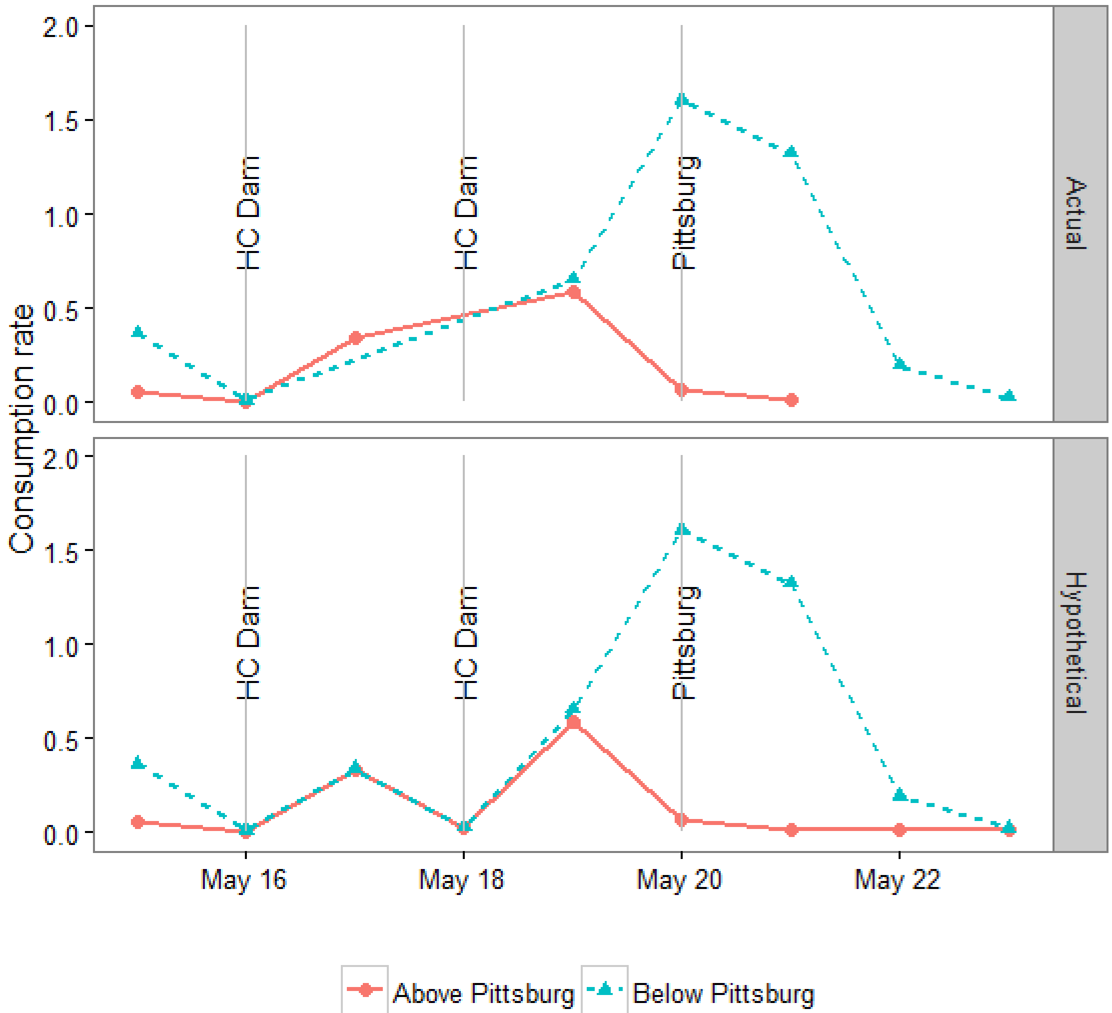

FIGURE 9. Actual (top panel) and hypothetical (bottom panel) daily consumption rate (fish/bass/day) of subyearling Chinook Salmon by Smallmouth Bass above and below Pittsburg Landing in the upper portion of the Snake River. Vertical bars represent dates of hatchery releases of subyearlings at Hells Canyon Dam and Pittsburg Landing. Hypothetical consumption is that which we expected if we had sampled every day following hatchery releases.

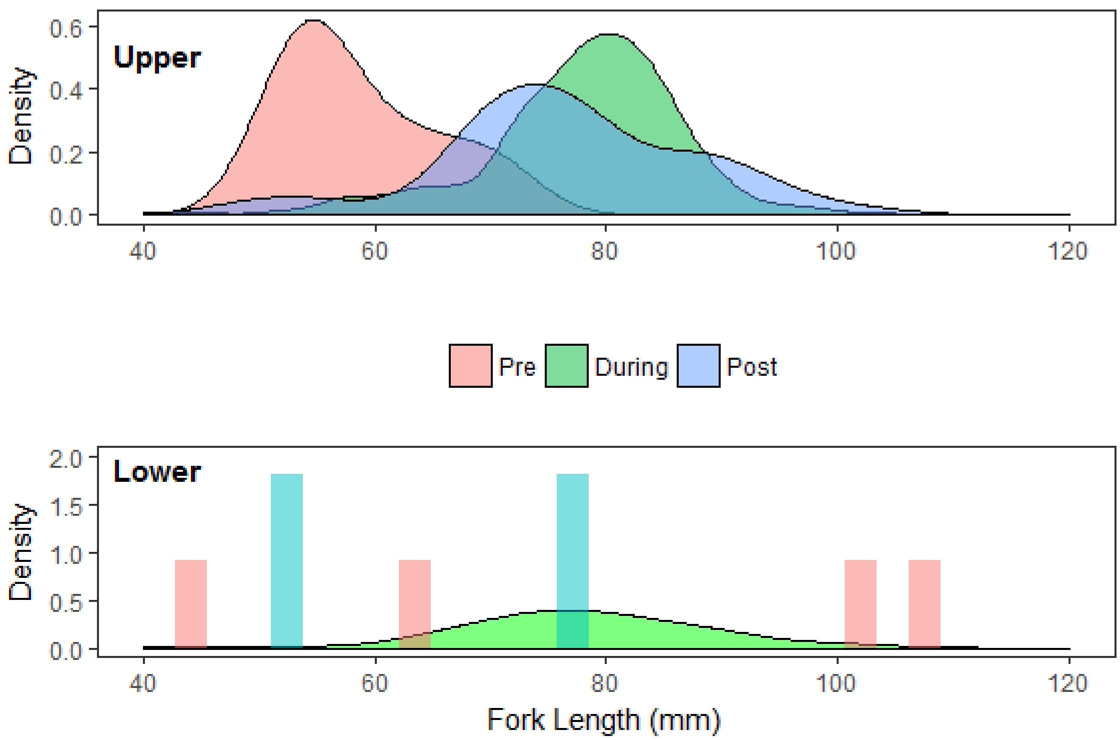

FIGURE 10. Size distribution of subyearling Chinook salmon consumed by smallmouth bass before (Pre), during, and after (Post) hatchery releases of subyearlings in the upper and lower reaches of the Snake River.

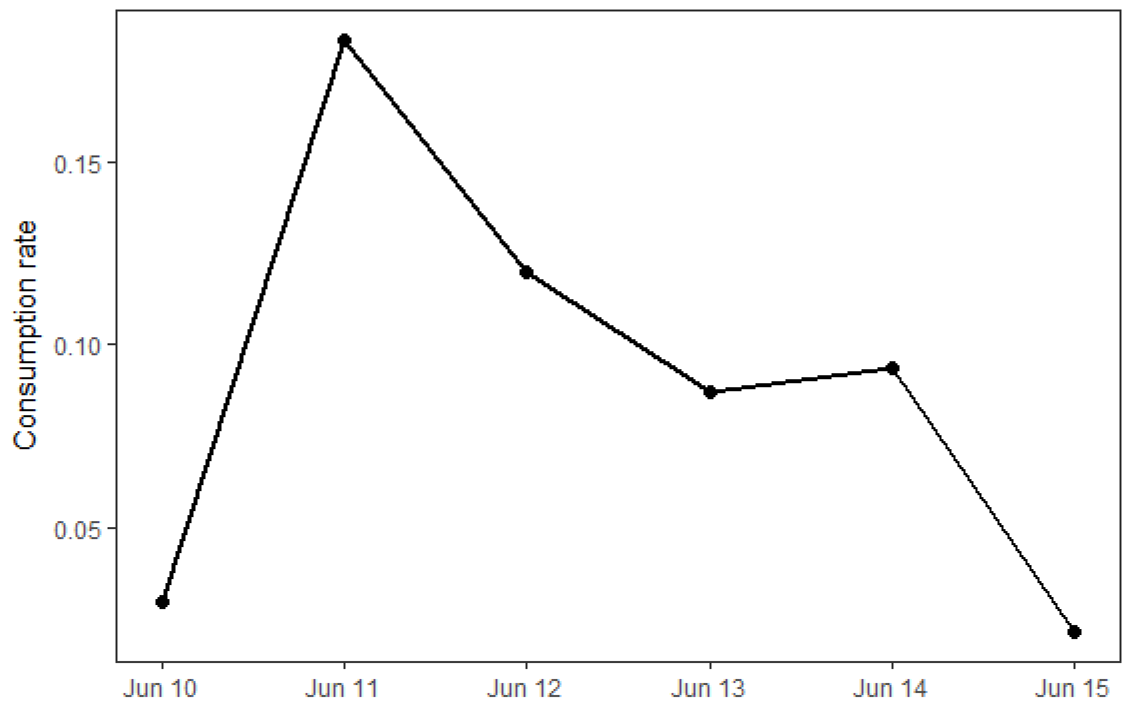

FIGURE 11. Smallmouth bass consumption rate (fish/bass/day) of subyearling Chinook Salmon following a hatchery release at Captain John acclimation facility in the lower portion of the Snake River, 2016.

Discussion/Conclusion

Attributes of Spawning in Riverine Habitat

The functional growth relations that we modeled, between total escapement and the number of shallow and deep sites used, were the first of several lines of evidence for density dependence in this report. The functional growth relation between total escapement and the numbers of shallow and deep redds counted provided growing but relatively weak evidence for density dependence as clear maximums were not attained. Together, the four curve fitting exercises for attributes of spawning showed that most of the suitable spawning habitat has been used by spawners, but the habitat has not reached redd capacity. In support of that conclusion, the maximum redd count observed in the Snake River reaches combined was 3,155 or 1,287 below the estimated redd capacity of 4,442 for the river (Groves et al. 2013). We are questioning the accuracy of our redd counts in recent years, however, because the large numbers of redds are difficult to count from the air especially when redds are superimposed. Superimposition occurs when a new redd is constructed on top of one or more previously constructed redds. In addition to being a source of counting error, that process can dislodge eggs in the previously constructed redds and be the underlying mechanism leading to the Beverton-Holt and Ricker-type recruitment we reported earlier.

In cooperation with staff of the Idaho Power Company who pioneered the use of Unmanned Aerial Systems (sUAS) for counting redds, we worked with statisticians from the University of Idaho to design and implement a study to improve monitoring of superimposition and accuracy of our redd counts. That effort came to fruition in 2015. For the present, superimposition is likely common in the Snake River reaches at the spawner escapement levels observed in recent years. In 2014, video taken from a sUAS was used to determine that some superimposition occurred at 43% of the sites used along the Snake River reaches, and that heavy superimposition occurred at 28% of the sites used (P. Groves, Idaho Power Company, unpublished data). Continued collection of redd count data will aid in monitoring and managing the recovery of the Snake River fall Chinook Salmon ESU.

Attributes of Natural-Origin Juveniles Rearing in Riverine Habitat and Emigrating through Lower Granite Reservoir

We found that recruitment of natural-origin fall Chinook Salmon subyearlings assessed based on CPUE declined at high total escapements (i.e., overcompensation or Ricker-recruitment). The three most plausible causes of that decline are redd superimposition, egg predation, and juvenile predation. As mentioned, superimposition has only been assessed in 2014. Additional analyses of video footage collected by UAS in earlier years as wells 2015 would be informative. White Sturgeon *Acipenser transmontanus* have been observed along the Snake River reaches hovering over redds in both underwater and sUAS video footage (P. Groves, Idaho Power Company, unpublished data). That footage provides circumstantial evidence for egg predation. Ongoing research by staff of project 199102900 (described later in this report) has also provided evidence that predation on natural-origin fall Chinook Salmon subyearlings by Smallmouth Bass has increased as total escapement of fall Chinook Salmon spawners has increased. Evidence for density dependent mechanisms, however, is not solely limited to the spawning life stage or predator/prey interactions.

Intra-specific density-dependent mechanisms during rearing in riverine habitat are also evident in this report. Growth rates of natural-origin parr in riverine rearing habitat declined slightly as parr abundance increased. Timing of downstream dispersal into the reservoir was also earlier and dispersal size decreased between abundance periods.

Our results on natural-origin subyearling smolts in Lower Granite Reservoir provided ample evidence for density dependence noting that we are questioning our 2015 estimate of annual passage abundance of natural-origin subyearling smolts. In addition to finding evidence for Beverton-Holt and Ricker-type recruitment, smolt growth declined between abundance periods in association with density-dependent changes in size at dispersal from riverine habitat, migrational behavior in the reservoir, and increased concentrations of subyearling smolts in the reservoir as the hatchery program expanded (Connor et al. 2013). In turn, time of passage through the reservoir of natural-origin smolts became earlier and smolt size decreased. During the low abundance period, natural-origin smolts also remained in the reservoir later into the summer, which allowed them to experience warmer temperatures and grow to larger sizes. However, during the high abundance period, natural-origin smolts spent more time migrating and less time lingering and feeding resulting in passage through the reservoir before they could benefit from warmer water temperatures that favored growth.

Attributes of Seasonal Variation in Smallmouth Bass Diets and Consumption of Subyearling Chinook Salmon during Rearing in Riverine Habitat

Not unexpectedly, Smallmouth Bass responded to releases of hatchery juvenile Chinook Salmon by preying heavily on them which was reflected in their consumption rates. Responses were much more pronounced in the upper portion of our study area where consumption increased to 1.60 subyearlings/bass/day. Nelle (1999) also found a similar predation response in the late 1990s when he evaluated Smallmouth Bass consumption following a release of hatchery subyearlings at Pittsburg Landing. He estimated a consumption rate of 1.143 subyearlings/bass/day for 150–249-mm bass following a release during July of 1997. Our estimates of consumption in upper Hells Canyon were probably conservative as angled bass often regurgitated numerous juvenile salmon as they were being reeled in and we could not account for this loss.

The predation response by Smallmouth Bass to a release of hatchery subyearlings is relatively short in Hells Canyon. We found that consumption rates typically returned to pre-release levels within a day or two, which may be due to the relatively short residence time of subyearlings in the river. Following release, hatchery fish tend to disperse downstream rapidly in the unimpounded Snake River in Hells Canyon (USFWS, unpublished), which probably shortens their exposure to predators. Thus, predation risk and loss is very high for a short time, but then quickly drops. However, natural-origin subyearlings rearing along shorelines would be exposed to predation for a longer time. Sampling logistics prevented us from sampling every day following each release which is why we speculated on the predation response on days following some hatchery releases (Figure 9). The best support for the assumption we made is shown by the response to the hatchery fish released at Hells Canyon Dam on May 18. Those fish would have travelled downstream to reach our study area by May 19 during with predation was likely high. On May 20, we observed low consumption rates suggesting that most hatchery fish had moved downstream. At a minimum, most consumption rates return to pre-release levels within 2 d of a hatchery release, but using a return to pre-release levels after 1 d results in more conservative loss estimates. Sampling at 1-d intervals in the future should help refine the pattern in consumption rate.

The pattern in Smallmouth Bass consumption rates was different in the lower portion of Hells Canyon. Below Captain John, bass consumption rates increased rapidly following the release of hatchery fish but never approached the rates observed in the upper canyon. Consumption rates also took longer to return to pre-release levels. There may be a number of reasons for these observations. First, other prey such as Sand Rollers *Percopsis transmontana*, which are absent in the upper canyon, may have reduced bass consumption of subyearlings. Second, there are fewer bass in the river below Captain John than in the upper canyon and this may have resulted in lower overall consumption rates. Finally, lingering hatchery fish from upstream releases that moved through the lower river may have contributed to the longer duration of elevated bass consumption rates.

Our consumption and loss estimates were probably influenced by spatial factors. The much higher consumption and loss estimates from the Pittsburg Landing release was most likely influenced by the study area being located immediately downstream of the release site. In contrast, the study reach above Pittsburg Landing was located 33.2 km downstream of Hells Canyon Dam. This spatial difference allowed fish to disperse more as they moved downstream from the dam which likely resulted in lower densities compared to fish released at Pittsburg Landing. We hypothesize that bass consumption rates decrease with increasing distance from release locations as the pulse of released subyearlings spreads out and as their numbers decrease from mortality. We also hypothesize that as migration slows further downriver that bass consumption rates remain elevated longer as shown by our results from the lower river. Since bass densities are similar between our two upper study reaches, and daily release numbers were likely similar, then there is some evidence to support our first hypothesis.

Qualitative examination of length frequency distributions suggest that most of the salmonids consumed in the upper portion of our study area 1–2 days after the release were hatchery subyearlings. This is reasonable since hatchery fish are probably more naïve of predators than natural-origin fish. We were unable to determine origin of consumed subyearlings from the June release in the lower river due to the fact that hatchery fish were already in the system before the release. However, we suspect that bass consumed mainly hatchery subyearlings following the release at Captain John.

Adaptive Management & Lessons Learned

Multistage life cycle models provide a powerful framework for understating how each life stage of a population contributes to population growth rate (Moussalli and Hilborn 1986; Brooks and Powers 2007). When used for simulation, multistage models allow the relative effects of density dependence at different life stages to be explored in the context of management actions including Fish Population RM&E, Hydrosystem RM&E, Harvest RM&E, Hatchery RM&E, Predation and Invasive Species Management RM&E. For example, by using a multistage model, Greene and Beechie (2004) found that the importance of habitat restoration to population recovery of Chinook Salmon depended on the mechanisms of density dependence affecting particular life stages.

Multistage models may also be used as an analytical framework to explicitly estimate demographic parameters of a population model. This approach has an advantage over single-stage stock-recruitment models by allowing population growth rates to be partitioned among life stages rather than aggregated over an entire life cycle. Such partitioning allows for estimating 1) stage-specific estimates of density dependence, and 2) stage specific effects of environmental factors or management actions. Zabel et al. (2005) estimated parameters of a multistage model used in the context of a population viability analysis spring/summer Chinook Salmon in the Snake River, but such an approach has yet to be applied to fall Chinook Salmon in the Snake River basin.

Typically, data informing estimates of abundance at particular "check points" in the life cycle dictates the complexity of multistage models that can be fit to data. For fall Chinook Salmon, we will start with a two-stage model that encompasses: 1) upstream passage of spawners at Lower Granite Dam to the subsequent the downstream passage of their progeny at the Dam, and 2) downstream passage of juveniles at Lower Granite Dam to their subsequent return from the ocean and passage at the Dam 3–5 years later. This approach partitions the life cycle of fall Chinook Salmon both spatially and temporally, which will allow us to fit and compare alternative models with covariates specific to each stage. We are building such a model while participating in the AMIP process where ideas on model development are shared among the members. Information is also presented at AFS meetings and regional forums to disseminate it and receive useful feedback from peers.

References

Beamesderfer, R. C. P., D. L. Ward, and A. A. Nigro. 1996. Evaluation of the biological basis for a predator control program on Northern Squawfish (Ptychocheilus oregonensis) in the Columbia and Snake rivers. Canadian Journal of Fisheries and Aquatic Sciences 53:2898–2908.

Brooks, E. N. and J. E. Powers. 2007. Generalized compensation in stock-recruit functions: properties and implications for management. ICES Journal of Marine Science 64: 413–424.

CBR (Columbia Basin Research). 2015. Columbia River DART (data access in real time). CBR, University of Washington, Seattle. Available: www.cbr.washington.edu/dart/dart.html. (June 2012).

Connor, W.P., H.L. Burge, R. Waitt, and T.C. Bjornn. 2002. Juvenile life history of wild fall Chinook Salmon in the Snake and Clearwater rivers. North American Journal of Fisheries 22:703–712.

Connor, W. P., H. L. Burge, J. R. Yearsley, and T. C. Bjornn. 2003b. The influence of flow and temperature on survival of wild subyearling fall Chinook Salmon in the Snake River. North American Journal of Fisheries Management 23:362–375.

Connor, W. P., S. G. Smith, T. Andersen, S. M. Bradbury, D. C. Burum, E. E. Hockersmith, M. L. Schuck, G. W. Mendel, and R. M. Bugert. 2004. Post-release performance of hatchery yearling and subyearling fall Chinook Salmon released into the Snake River. North American Journal of Fisheries Management 24:545-560.

Connor, W. P., K. F. Tiffan, J., J. M. Plumb, and C. M. Moffitt. 2013. Evidence for density – dependent changes in growth, downstream movement, and size of Chinook Salmon subyearlings in a large river landscape. Transactions of the American Fisheries Society 142:1453–1468.

Connor, W. P., F. L. Mulllins, K. F. Tiffan, R. W. Perry, J. M. Erhardt, S. J. St. John, B. K. Bickford, and T. N. Rhodes. 2015. Research, Monitoring, and Evaluation of Emerging Issues and Measures to Recover the Snake River Fall Chinook Salmon ESU. 2014 Annual Report to the Bonneville Power Administration, Project 199102900, Portland, Oregon.

Fritts, A.L., and T.N. Pearsons. 2004. Smallmouth Bass predation on hatchery and wild Salmonids in the Yakima River, Washington. Transactions of American Fisheries Society 133:880-895.

Greene, C. M. and T. J. Beechie. 2004. Consequences of potential density-dependent mechanisms on recovery of ocean-type Chinook Salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Sciences 61: 590–602.

Groves, P.A., J.A. Chandler, B. Alcorn, T.J. Richter, W.P. Connor, A.P. Garcia, and S.M. Bradbury. 2013. Evaluating Salmon spawning habitat capacity using redd survey data. North American Journal of Fisheries Management 33:707–716.

Groves, P. A., and J. A. Chandler. 1999. Spawning habitats used by fall Chinook Salmon in the Snake River. North American Journal of Fisheries Management 19:912–922.

Groves, P. A, B. Alcorn, B., M. M. Wiest, J. M., Maselko, and W. P. Connor. 2016. Testing unmanned aircraft systems (UASs) for salmon spawning surveys. FACETS. DOI: 10.1139/facets-2016-0019. http://www.facetsjournal.com/article/facets-2016-0019/. (Available November 2016).

Hansel, H.C., S.D. Duke, P.T. Lofy, and G.A. Gray. 1988. Use of diagnostic bones to identify and estimate original lengths of ingested prey fishes. Transactions of American Fisheries Society 117:55-62.

ISAB (Independent Scientific Advisory Board). 2015. Density dependence and its implications for fish management and restoration programs in the Columbia River basin. ISAB 2015-1. Available: http://www.nwcouncil.org/media/7148891/isab2015-1.pdf (March 2015).

Moussalli, E. and R. Hilborn. 1986. Optimal stock size and harvest rate in multistage life history models. Canadian Journal of Fisheries and Aquatic Sciences 43: 135–141.

Naughton, G.P., D.H. Bennett, and K.B. Newman. 2004. Predation on juvenile Salmonids by Smallmouth Bass in the Lower Granite Reservoir system, Snake River. North American Journal of Fisheries Management 24:534-544.

Nelle, R.D. 1999. Smallmouth Bass predation on juvenile fall Chinook Salmon in the Hells Canyon Reach of the Snake River, Idaho. Master’s Thesis, University of Idaho, Moscow.

Parrish, J.K., K. Haapa-aho, W. Walker, M. Stratton, J. Walsh, and H. Ziel. 2006. Small-bodied and juvenile fishes of the Mid-Columbia Region including keys to diagnostic otoliths and cranial bones. Draft Version, March 2006. University of Washington, Seattle WA. 137 p.

Peters, C. N., D. R. Marmorek, and R. B. Deriso. 2001. Application of decision analysis to evaluate recovery actions for threatened Snake River fall Chinook Salmon (Oncorhynchus tshawytscha). Canadian Journal of Fisheries and Aquatic Science 58:2447–2458.

Prentice, E. F., T. A. Flagg, and C. S. McCutcheon. 1990a. Feasibility of using implantable passive integrated transponder (PIT) tags in Salmonids. American Fisheries Society Symposium 7:317–322.

Rainey, S.R., L.A. Reese, and T.O. Wik. 2006. Removable spillway weir for fish passage, power, and water quality benefits. Hydro Review 25:118-124.

Rogers, J.B., and C.C. Burley . 1991. A sigmoid model to predict gastric evacuation rates of Smallmouth Bass (Micropterus dolomieu) fed juvenile Salmon. Canadian Journal of Fisheries and Aquatic Sciences 48:933-937.

Rieman, B.E., R.C. Beamsderfer, S. Vigg, and T.P. Poe. 1991. Estimated loss of juvenile Salmonids to predation by Northern Squawfish, Walleyes, and Smallmouth Bass in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120:448-458.

Shields, P. A. and S. R. Carlson. 1996. Effects of formalin and alcohol preservation on lengths and weights of juvenile sockeye salmon. Alaska Fishery Research Bulletin 3:81-93.

Tabor, R.A., R.S. Shively, and T.P. Poe. 1993. Predation on juvenile Salmonids by Smallmouth Bass and Northern Squawfish in the Columbia River near Richland, Washington. North American Journal of Fisheries Management 13:831-838.

Tiffan, K. F., and W. P. Connor. 2011. Distinguishing between natural and hatchery Snake River fall Chinook Salmon subyearlings in the field using body morphology. Transactions of the American Fisheries Society 140:21–30.

Vigg, S., T.P. Poe, L.A. Prendergast, and H.C. Hansel. 1991. Rates of consumption of juvenile Salmonids and alternate prey fish by Northern Squawfish, Walleyes, Smallmouth Bass, and Channel Catfish in John Day Reservoir, Columbia River. Transactions of the American Fisheries Society 120:421-438.

Ward, D. L., J. H. Petersen, and J. J. Loch. 1995. Index of predation on juvenile salmonids by northern squawfish in the lower and middle Columbia River and in the lower Snake River. Transactions of the American Fisheries Society 124:321-334.

Zabel, R. W., M. D. Scheuerell, M. M. McClure, and J. G. Williams. 2006. The interplay between climate variability and density dependence in the population viability of Chinook Salmon. Conservation Biology 20:190–200.Fisheries Review 39:10–14.

Zimmerman, M.P. 1999. Food habits of Smallmouth Bass, Walleyes, and Northern Pikeminnow in the lower Columbia River Basin during outmigration of juvenile anadromous Salmonids. Transactions of the American Fisheries Society 128:1036-1054.

Appendices

A.1: Data Links

Redd count data: Available:http://www.fpc.org/
Seining data : Available: http://www.streamnet.org/
PIT-tag data: Available:http://www.ptagis.org/

Other data is backed up on site and remotely. Inquires can be sent to william_connor@fws.gov or ktiffan@usgs.gov.

A.2: Bibliography of Published Journal Articles (N = 48)

Bennett, D.H., W.P. Connor, and C.A. Eaton. 2003. Substrate composition and emergence success of fall Chinook Salmon in the Snake River. Northwest Science 77:93–99.

Connor, W.P., H.L. Burge, and D.H. Bennett. 1998. Detection of subyearling Chinook Salmon at a Snake River dam: Implications for summer flow augmentation. North American Journal of Fisheries Management 18:530–536.

Connor, W.P., R.K. Steinhorst, and H.L. Burge. 2000. Forecasting survival and passage for migratory juvenile Salmonids. North American Journal of Fisheries Management 20:650–659.

Connor, W.P. and several coauthors. 2001. Early life history attributes and run composition and of wild subyearling Chinook Salmon recaptured after migrating downstream past Lower Granite Dam. Northwest Science 75:254–261.

Connor, W.P, A.R. Marshal, T.C. Bjornn, and H.L. Burge. 2001. Growth and long-range dispersal by wild subyearling spring and summer Chinook Salmon in the Snake River. Transactions of the American Fisheries Society 130:1070–1076.

Connor, W.P., A. P. Garcia, A. H. Connor, E. O. Garton, P. A. Groves, and J. A. Chandler. 2001. Estimating the carrying capacity of the Snake River for fall Chinook Salmon redds. Northwest Science 75:363–370.

Connor, W.P., H.L. Burge, R. Waitt, and T.C. Bjornn. 2002. Juvenile life history of wild fall Chinook Salmon in the Snake and Clearwater rivers. North American Journal of Fisheries 22:703–712.

Connor, W.P., H.L. Burge, J.R. Yearsley, and T.C. Bjornn. 2003. The influence of flow and temperature on survival of wild subyearling fall Chinook Salmon in the Snake River. North American Journal of Fisheries Management 23:362–375.

Connor, W.P., R.K. Steinhorst, and H.L. Burge. 2003. Migrational behavior and seaward movement of wild subyearling fall Chinook Salmon in the Snake River. North American Journal of Fisheries Management 23:414–430.

Connor, W.P., and H.L. Burge. 2003. Growth of wild subyearling Chinook Salmon in the Snake River. North American Journal of Fisheries Management 23:594–599.

Connor, W.P., C.E. Piston, and A.P. Garcia. 2003. Temperature during incubation as one factor affecting the distribution of Snake River fall Chinook Salmon spawning areas. Transactions of the American Fisheries Society 132:1236–1243.

Connor, W.P., S.G. Smith, T. Andersen, S.M. Bradbury, D.C. Burum, E.E. Hockersmith, M.L. Schuck, G.W. Mendel, and R.M. Bugert. 2004. Post-release performance of hatchery yearling and subyearling fall Chinook Salmon released into the Snake River. North American Journal of Fisheries Management 24:545–560.

Connor, W.P., J.G. Sneva, K.F. Tiffan, R.K. Steinhorst, and D. Ross. 2005. Two alternative juvenile life histories for fall Chinook Salmon in the Snake River basin. Transactions of the American Fisheries 134:291–304.

Connor, W. P., and A. P. Garcia. 2006. Pre-spawning movement of wild and hatchery fall Chinook Salmon in the Snake River. Transactions of the American Fisheries Society 135:131–139.

Connor, W.P., and K.F. Tiffan. 2012. Evidence for parr growth as a factor affecting parr-to-smolt survival. Transactions of the American Fisheries Society 141:1207–1218.

Connor, W. P., K. F. Tiffan, J., J. M. Plumb, and C. M. Moffitt. 2013. Evidence for density – dependent changes in growth, downstream movement, and size of Chinook Salmon subyearlings in a large river landscape. Transactions of the American Fisheries Society 142:1453–1468.

Dauble D., R.L. Johnson, and A.P. Garcia. 1999. Fall Chinook Salmon spawning in the tailraces of hydroelectric projects. Transactions of the American Fishery Society 128:672–679.

Erhardt, J.M., and K.F. Tiffan. 2016. Ecology on nonnative Siberian prawn (*Palaemon modestus*) in the lower Snake River, Washington, USA. Aquatic Ecology 50:607-621.

Garcia, A.P., W.P. Connor, D.J. Milks, S.J. Rocklage, and R.K. Steinhorst. 2004. Movement and spawner distribution of hatchery fall Chinook Salmon adults acclimated and released as yearlings at three locations in the Snake River basin. North American Journal of Fisheries Management 24:1134–1144.

Garland, R.D., K.F. Tiffan, D.W. Rondorf, and L.O. Clark. 2002. Comparison of subyearling fall Chinook Salmon's use of riprap revetments and unaltered habitats in Lake Wallula of the Columbia River. North American Journal of Fisheries Management 22:583–589.

Groves, P.A., and A.P. Garcia. 1998. Two carriers used to suspend an underwater video camera from a boat. North American Journal of Fisheries Management 18:1004–1007.

Groves, P.A., J.A. Chandler, B. Alcorn, T.J. Richter, W.P. Connor, A.P. Garcia, and S.M. Bradbury. 2013. Evaluating Salmon spawning habitat capacity using redd survey data. North American Journal of Fisheries Management 33:707–716.

Groves, P. A, B. Alcorn, B., M. M. Wiest, J. M., Maselko, and W. P. Connor. 2016. Testing unmanned aircraft systems (UASs) for salmon spawning surveys. FACETS. DOI: 10.1139/facets-2016-0019. http://www.facetsjournal.com/article/facets-2016-0019/. (Available November 2016).

Haskell, C.A., K.F. Tiffan, and D.W. Rondorf. 2006. Food habits of juvenile American shad and dynamics of zooplankton in the lower Columbia River. Northwest Science 80:47–64.

Haskell, C.A., R.D. Baxter, and K.F. Tiffan. 2006. Range expansion of an exotic Siberian prawn to the Lower Snake River. Northwest Science 80:311–316

Haskell, C.A., K.F. Tiffan, and D.W. Rondorf. 2013. The effects of juvenile American shad planktivory on zooplankton production in Columbia River food webs. Transactions of the American Fisheries Society 142:606–620.

Marshall, A.R., H.L. Blankenship, and W.P. Connor. 2000. Genetic characterization of naturally spawned Snake River fall-run Chinook Salmon. Transactions of the American Fisheries Society 129:680–698.

Plumb, J.P., W.P. Connor, K.F. Tiffan, C.M. Moffitt, R.W. Perry, and N.S. Adams. 2012. Estimating and predicting collection probability of fish at dams using multistate modeling. Transactions of the American Fisheries Society 141:1364–1373.

Rasmussen, C. R., C. O. Ostberg, D. R. Clifton, J. L. Holloway, and R. J. Rodriguez. 2003. Identification of a genetic marker that discriminates ocean-type and stream-type Chinook Salmon in the Columbia River basin. Transactions of the American Fisheries Society 132:131–142.

Rosenberger, S.J., W.P Connor, C.A. Peery, D.J. Milks, M.L. Shuck, J.A. Hesse, and S.G. Smith. 2013. Acclimation enhances postrelease performance of hatchery fall Chinook Salmon subyearlings while reducing the potential for interaction with natural fish. North American Journal of Fisheries Management 33:519–528.

Smith, S.G., W.D. Muir, E.E. Hockersmith, R.W. Zabel, R.J. Graves, C.V. Ross, W.P. Connor, and B.D. Arnsberg. 2003. Influence of river conditions on survival and travel time of Snake River subyearling fall Chinook Salmon. North American Journal of Fisheries Management 23:939–961.

Tiffan, K.F., D.W. Rondorf, and P.G. Wagner. 2000. Physiological development and migratory behavior of subyearling fall Chinook Salmon in the Columbia River. North American Journal of Fisheries Management 20:28–40.

Tiffan, K.F., D.W. Rondorf, R.D. Garland, and P.A. Verhey. 2001. Identification of juvenile fall versus spring Chinook Salmon migrating through the lower Snake River based on body morphology. Transactions of the American Fisheries Society 129:1389–1395.

Tiffan, K.F., R.D. Garland, and D.W. Rondorf. 2002. Quantifying flow-dependent changes in subyearling fall Chinook rearing habitat and stranding area using two-dimensional spatially-explicit modeling. North American Journal of Fisheries Management 22:713–726.

Tiffan, K.F., C.A. Haskell, and D.W. Rondorf. 2003. Thermal exposure of juvenile fall Chinook Salmon migrating through a lower Snake River Reservoir. Northwest Science 77:100–109.

Tiffan, K.F., L.O. Clark, R.D. Garland, and D.W. Rondorf. 2006. Variables influencing the presence of subyearling fall Chinook Salmon in shoreline habitats of the Hanford Reach, Columbia River. North American Journal of Fisheries Management 26:351–360.

Tiffan, K.F., T.J. Kock, W.P. Connor, R.K. Steinhorst, and D.W. Rondorf. 2009. Behavioural thermoregulation by subyearling fall ± autumn) Chinook Salmon Oncorhynchus tshawytscha in a reservoir. Journal of Fish Biology 74:1562–1579.

Tiffan, K.F., T.J. Kock, C.A. Haskell, W.P. Connor, and R.K. Steinhorst. 2009. Water velocity, turbulence, and migration rate of subyearling fall Chinook Salmon in the free-flowing and impounded Snake River. Transactions of the American Fisheries Society 138:373–384.

Tiffan, K.F., and W.P. Connor. 2011. Distinguishing between natural and hatchery Snake River fall Chinook Salmon subyearlings in the field using body morphology. Transactions of the American Fisheries Society 140:21–30.

Tiffan, K.F., T.J. Kock, W.P. Connor, F. Mullins, and R.K. Steinhorst. 2012. Downstream movement of fall Chinook Salmon juveniles in the lower Snake River reservoirs during winter and early spring. Transaction of the American Fisheries Society 141:285–293.

Tiffan, K. F., J. M. Erhardt, and S. J. St. John. 2014. Prey availability, consumption, and quality contribute to variation in growth of subyearling Chinook Salmon rearing in riverine and reservoir habitats. Transactions of the American Fisheries Society 143:219–229.

Tiffan, K. F., R. W. Perry, W. P. Connor, F. L. Mullins, C. D. Rabe, and D. D. Nelson. 2015. Survival, growth, and tag retention in age-0 Chinook salmon implanted with 8-, 9-, and 12-mm PIT tags. North American Journal of Fisheries Management 35:845–852.

Tiffan, K.T., J.R. Hatten, and D.A. Trachtenbarg. 2016. Assessing juvenile salmon rearing habitat and associated predation risk in a lower Snake River reservoir. River Research and Applications 32:1030-1038.

Tiffan, K.F., and W.R. Hurst. 2016. Feeding ecology of non-native Siberian prawns Palaemon modestus (Heller, 1862) (Decapoda, Palaemonidae) in the lower Snake River, Washington, USA. Crustaceana 89:721-736.

Tiffan, K.F., J.M. Erhardt, and B.K. Bickford. In press. Ecology of the opossum shrimp (Neomysis mercedis) in a lower Snake River reservoir, Washington. Northwest Science.

Tiffan, K.F., J.M. Erhardt, T.N. Rhodes, and R.J. Hemingway. In press. Ecology of the Sand Roller (*Percopsis transmontana*) in a lower Snake River reservoir, Washington. Northwestern Naturalist.

Venditti, D.A., D.W. Rondorf, J.M. Kraut. 2000. Migratory behavior and forebay delay of radio-tagged juvenile fall Chinook Salmon in a lower Snake River impoundment. North American Journal of Fisheries Management 20:41–52.

Williams, J.G., Zabel, R.S., Waples, R.S., Hutchings, J.A., and Connor, W.P. 2008. Potential for anthropogenic disturbances to influence evolutionary change in the life history of a threatened Salmonid. Evolutionary Applications doi:10.1111/j.1752–4571.2008.00027.x: 271-285.

A.3: Additional Research

Detectability of 8-mm, 9-mm, and 12-mm PIT Tags implanted in juvenile Chinook Salmnon at Lower Granite Dam

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Introduction

The ability to represent a population of migratory juvenile fish with Passive Integrated Transponder (PIT) tags becomes difficult when the minimum tagging size is larger than the average size at which fish begin to move downstream. The current minimum recommended size for tagging juvenile salmonids with 12-mm PIT tags in the Columbia River basin is ≥65 mm FL. From 1992 to1999, the mean FL of subyearling Chinook Salmon from the Snake River upstream of Lower Granite Dam was 65 mm ± 1 mm (95% CI) at initial downstream dispersal, but subsequently decreased to 55 ± 1 mm from 2000 to 2011 during the same dispersal period (Connor et al. 2013). Tags that are smaller (e.g., 8 and 9 mm) than the commonly used 12-mm PIT tag are now currently available. Tiffan et al. (2015) found that 40–49-mm subyearling Chinook salmon could be implanted with 8-mm and 9-mm tags without largely affecting growth and survival in the laboratory, and recommended experimental use of the smaller tags for subyearlings in the field. The use of smaller PIT tags (i.e., 8 and 9 mm) would allow researchers to more fully represent the juvenile Snake River fall Chinook Salmon population that undergoes size-related dispersal, as well as reducing tag burden-related effects (Tiffan et al. 2015).

Many PIT-tagged juvenile salmonids in the Snake and Columbia river basins are detected at mainstem dams to provide various population, behavioral, and survival metrics. The detectability of 8-mm tags at mainstem dams has not been evaluated in juvenile salmonids, so it is unknown whether reliable detection information can be obtained from fish implanted with these tags. In 2016, we conducted a simple test of detectability of 8-mm, 9-mm, and 12-mm PIT tags at Lower Granite Dam. The objective of the test was to determine the detection efficiency of the different tags implanted in subyearling fall Chinook salmon and released within the fish bypass system.

Methods

We evaluated the detectability of four different PIT tags: 8-mm, Biomark; 8-mm, Oregon RFID; 9-mm, Biomark; and 12-mm, Biomark (Table 1). On June 7, Smolt Monitoring Program personnel at Lower Granite Dam collected approximately 300 untagged subyearling Chinook salmon and held them in four 50-gallon containers supplied with river water. The next day, we tagged fish systematically by implanting the first fish with an 8-mm Biomark tag, the second fish received an 8-mm Oregon RFID tag, the third fish received a 9-mm tag, and the fourth fish received a 12-mm tag. This cycle was then repeated until 75 fish were tagged with each tag type. PIT tagging and anesthetization followed the recommendations of Prentice et al. (1990a, 1990b) and the PIT-tag Steering Committee (PTSC 2014) with some exceptions stated below.

Tagging proceeded by first anesthetizing fish in 18.9 L of oxygenated water that contained 5 mL of MS-222 in solution at a concentration of 100 mg/L, 0.5 g $NaHC0_3$ as a buffer, and 2.5 mL of Polyaqua as a replacement "slime" mucoprotein coating (Connor et al. 1998). A fish was then randomly selected and weighed to the nearest 0.1 g, measured (FL) to the nearest mm, and tagged using an injector. Our tagging apparatus was a 10-cc syringe body affixed with 12-gauge needle for use with 9- and 12-mm tags or a 14-gauge needle for use with 8-mm tags. Each syringe body had a slot cut into the side that was approximately 10 × 3 mm that allowed a thumb screw to be inserted into a plastic cylinder contained within the syringe that had an attached push rod that extended into the barrel of the needle. PIT tags were inserted into the beveled end of the needle and then inserted into the fish by moving the thumb screw toward the end of the needle. Sedated fish were placed head first and dorsal surface down into the corner of a wet, notched sponge leaving only a slight portion of the ventral and posterior portion of the body exposed. The syringe was then held parallel to the fish and a slight downward pressure was applied to the needle to make a small incision (large enough only to accommodate the insertion of the needle bevel) off the ventral midline. The tag was then inserted into the fish's peritoneal cavity while the hands and forearms of the tagging person were braced for stability (Tiffan et al. 2015). All tagging information, including the presence of coded-wire tags and additional marks, were recorded using P3 software and uploaded to the PIT Tag Information System (PTAGIS 2016).

After tagging, fish were allowed to recover for 15 min in 18.9 L of oxygenated water with 2.5 mL of Polyaqua solution and then transferred to one of four 75-gallon replicate circular tanks supplied with river water. Each tank contained equal numbers of fish tagged with each tag size and type. Tagged fish were held for an additional 20–24 h to check for shed tags and mortalities. On the day of release, groups of 20 fish were transferred to aerated 5-gallon buckets and released into the upwell area of the bypass system upstream of the wet separator. Ten fish were released every 5 min to minimize the chance of tag detection "collisions" and to maximize individual detections. This resulted in approximately 30 releases of 10 fish each over two and half hours. The release location exposed fish to detection by up to six PIT tag antennas located throughout the bypass system depending on the route they took (Figure 1). The first route would be across the separator to the adult return where fish could be detected by two antennas (F1 and F2; Figure 1). A second route would be through the separator gate antennas (either A1 and A2 or B1 and B2) and through three additional antennas (51, 52, and 53) to the sample tank. Fish taking this route could be detected by a maximum of five antennas. The final route would be through the separator gate antennas (either A1 and A2 or B1 and B2) and then through four additional antennas (C1, C2, 91, and 92) enroute to exiting to the river (Figure 1). Fish taking this route could be detected by a maximum of six antennas.

Table 1. Specifications of PIT tags implanted in subyearling fall Chinook Salmon at Lower Granite Dam in 2016.

Tag size (mm)	Model	Manufacturer	Weight (mg)	Length (mm)	Diameter (mm)
8	684293-003	Oregon RFID, Portland, OR	30	8.24	1.40
8	MiniHPT8	Biomark, Boise, ID	30	8.43	1.40
9	TXP148511B	Biomark, Boise, ID	66	8.91	2.10
12	TXP1411SST	Digital Angel, South St. Paul, MN	104	12.37	2.02

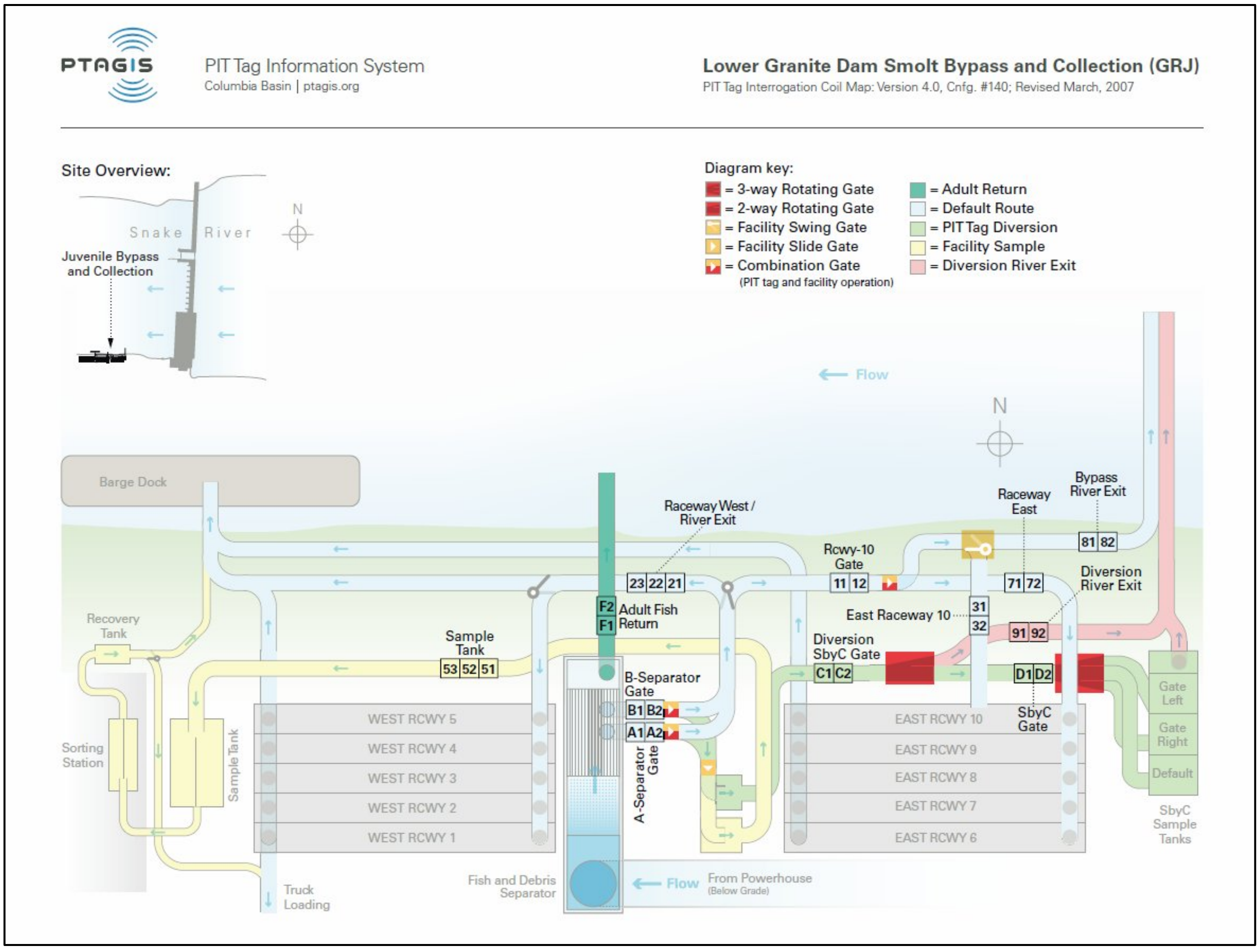

Figure 1. Map of PIT-tag interrogation system at Lower Granite Dam. Fish were released upstream of the upwell (the blue circle).

Results

We PIT tagged 304 fish, of which nine fish died after 24 h. No tags were expelled during the 24-h post-tagging holding period. Tagged fish lengths ranged from 57 to 128 mm and weights ranged from 1.4 to 22.4 g (Table 2). We released 295 fish into the bypass system for subsequent detection which comprised 74 fish tagged with 8-mm Biomark tags, 72 fish tagged with 8-mm Oregon RFID tags, 75 fish tagged with 9-mm Biomark tags, and 74 fish tagged with 12-mm Biomark tags. Detection rate of all tag sizes combined was 99.7%. Only one tagged fish was not detected, and it was tagged with an 8-mm Biomark tag.

From 98.6 to 100% (depending on tag type) of tagged fish were detected on at least two antennas (Table 3). The fish detected on only two antennas comprised five individuals that passed into the adult return, thus bypassing further detection opportunities. Four fish were detected passing into the sample tank; two of which were detected on four antennas and two of which were detected on all five antennas. Between 76.4 and 98.7% of the remaining 285 fish were detected on three, or more, of the possible six antennas (Table 3; Figure 1). In general, the 8-mm Oregon RFID tags were detected at slightly lower rates than the Biomark tags (Table 3).

Discussion

Total detection rate of combined tag groups was high (99.7%) with only one tag (an 8-mm Biomark tag) not being detected in this efficiency test. In addition, relatively high percentages (>76) of tagged fish were detected on all six PIT tag antennas. Even the few fish that passed through the adult return were detected on both antennas located there. These results are encouraging because they suggest that subyearlings implanted with 8-mm PIT tags should have detection efficiencies at Lower Granite Dam that are comparable to 9-mm and 12-mm tags. This also means that field use of 8-mm PIT tags should be practical when subsequent detection at Lower Granite Dam is of interest.

The mortality rate of post-tagged fish was relatively high at 3% (9 of 304 fish) and was likely due to stress related from collection, longer holding periods, tagger experience, and relatively high water temperatures (~17°C). All fish that died were >84 mm and 5 g. Tagging-related mortality was not too concerning for this study because we were primarily interested in using surviving fish as "vehicles" to carry tags through the bypass system. Mortality of much smaller fish tagged in the field would be of greater concern but has not been evaluated. However, 8-mm PIT tags may have advantages over larger tags. During field tagging in 2016, field personnel noted that the size of the incision associated with 14-gauge needles used to implant 8-mm tags was smaller than that made by 12-gauge needles used to implant 9-mm and 12-mm tags. Furthermore, we observed that fish recaptured in the field that were tagged with 8-mm tags (using 14-gauge needles) had healed more quickly than fish tagged with larger tags and needles. This might facilitate tag retention, reduce recovery time, and possibly reduce tagging mortality in the field.

Table 2. Mean±SD lengths and weights of subyearling fall Chinook salmon tagged at Lower Granite Dam to evaluate detection efficiency of different PIT tags.

Tag size (mm) and brand	FL (mm)	FL range (mm)	WT (g)	WT (g) range
8 Biomark	105±7.6	80-128	9.8±7.4	5.2-19.0
8 Oregon RFID	103±11.3	57-126	10.8±9.3	1.4-18.2
9 Biomark	104±8.7	79-122	10.8±2.5	4.4-16.1
12 Biomark	104±9.5	62-127	11.2±10.3	1.8-22.4

Table 3. Percent of PIT-tag groups detected on unique antennas in the bypass system at Lower Granite Dam.

Tag size (mm) and brand	≥2 antennas	≥3 antennas	≥4 antennas	≥5 antennas	6 antennas
8 Biomark	98.6	98.6	98.6	97.3	86.5
8 Oregon RFID	100.0	97.2	95.8	90.3	76.4
9 Biomark	100.0	98.7	98.7	92.0	77.3
12 Biomark	100.0	98.6	98.6	97.3	95.9

References

Connor, W.P., H.L. Burge, and D.H. Bennett. 1998. Detection of subyearling Chinook salmon at a Snake River dam: Implications for summer flow augmentation. North American Journal of Fisheries Management 18:530-536.

Connor, W.P., K.F. Tiffan, J.M. Plumb, and C.M. Moffitt. 2013. Evidence for density-dependent changes in growth, downstream movement, and size of Chinook salmon subyearlings in a large-river landscape. Transactions of the American Fisheries Society 142:1453-1468.

PTAGIS (PIT Tag Information System). 2016. Available at http:\\www.ptagis.org.

PTSC (PIT Tag Steering Committee). 2014. PIT Tag marking procedures manual, version 3.0.

Prentice, E.F., T.A. Flagg, and C.S. McCutcheon. 1990a. Feasibility of using implantable passive integrated transponder (PIT) tags in salmonids. Pages 317–322 *in* N.C. Parker, A.E. Giorgi, R.C. Heidinger, D.B. Jester, Jr., E.D. Prince, and G.A. Winans, editors. Fish-marking techniques. American Fisheries Society, Symposium 7, Bethesda, Maryland.

Prentice, E.F., T.A. Flagg, C.S. McCutcheon, and D.F. Brastow. 1990b. PIT-tag monitoring systems for hydroelectric dams and fish hatcheries. Pages 323– 334 *in* N.C. Parker, A.E. Giorgi, R.C. Heidinger, D.B. Jester, Jr., E.D. Prince, and G.A. Winans, editors. Fish-marking techniques. American Fisheries Society, Symposium 7, Bethesda, Maryland.

Tiffan, K.F., R.W. Perry, W.P. Connor, F.L. Mullins, C.D. Rabe, and D. Nelson. 2015. Survival, growth and tag retention in age-0 Chinook salmon implanted with 8-mm, 9-mm, and 12-mm PIT tags. North American Journal of Fisheries Management 35:845-852.